PB80-158132

Evaluation of Mound Systems for Purification of Septic Tank Effluent

Wisconsin Univ -Madison Dept of Soil Science

Prepared for

Office of Water Research and Technology
Washington, DC

1979

U.S. Department of Commerce
National Technical Information Service
NTIS

PB80-158132

1425.01

29) EVALUATION OF MOUND SYSTEMS FOR PURIFICATION OF SEPTIC TANK EFFLUENT,

BY

CHARLES FITZGERALD

A thesis submitted in partial fulfillment
of the requirements for the degree of

MASTER OF SCIENCE
(Soil Science Department)

at the
3a) UNIVERSITY OF WISCONSIN, 180p.

3b) 1979.

OWRT OFFICIAL
FILE COPY

B-099-WISC (2)

REPRODUCED BY
NATIONAL TECHNICAL
INFORMATION SERVICE
U.S. DEPARTMENT OF COMMERCE
SPRINGFIELD, VA. 22161

SELECTED WATER RESOURCES ABSTRACTS

INPUT TRANSACTION FORM

1. *Report No.*

2.

3. *Accession No.* **W** 80-04112

4. *Title*
Evaluation of Mound Systems for Purification of Septic Tank Effluent, (21 cop)

5. *Report Date*

6.

7. *Author(s)*
C[redacted] J. Fitzgerald

8. *Performing Organization Report No.*

9. *Organization*
Department of Soil Science [redacted]
University of Wisconsin-Madison ✓

10. *Project No.*
OWRT B-099-WIS (2)

11. *Contract/Grant No.*
14-34-0001-7200

12. *Sponsoring Organization*

13. *Type of Report and Period Covered*

15. *Supplementary Notes*

M.S. Thesis, 1979. 310 p., 26 fig~~s~~., 18 tab~~les~~, 112 ref~~s~~.

16. *Abstract*

Thirty-three mound systems for on-site disposal of septic tank effluent (STE) installed by commercial contractors in Wisconsin were surveyed and monitored for. ~~2-3 years to test whether they are being~~ proper~~ly~~ design~~ed~~, install~~ed~~ation, and operat~~ed~~ion, and ~~are achieving adequate~~ wastewater purification. Every system examined was correctly sized and constructed but many systems were being dosed inappropriately with large volumes of STE. In these systems, bacteria and nutrients (N and P) from the STE penetrate deeper into the soil than in systems being properly dosed with more frequent smaller volumes of STE. However all systems removed bacteria, viruses and phosphorus effectively and reduced the N load passing through the soil by nitrification-denitrification. No evidence of contamination by bacteria, N or P could be found in high groundwater near 4 systems located over permeable soils. The pressure system used for STE distribution in Wisconsin mounds prevents the formation of biological crusts where the STE enters the soil, a phenomenon which leads to failure of conventional systems. Three cases of mound system failure reported in the state were ~~found to be~~ due to faulty construction, not improper design or clogging. Wisconsin mound systems are expected to outlive and out perform conventional septic systems. (Harkin - Wis)

17a. *Descriptors* (*Efficiencies, *Wisconsin,)
*Septic Tanks, *Domestic Wastes, *Sewage Bacteria, *Sewage Disposal, *Sewage Treatment, *Soil Disposal Fields, *Water Pollution Source, *Coliforms, *Streptococcus, *Pseudomonas, *Viruses, *Nutrient Removal, *Public Health,

~~17b. Identifiers~~

Alternative septic systems, *mound systems, environmental sanitation, *water purification,

17c. *COWRR Field & Group* 05D

18. *Availability*
Available from the National Technical Information Service as PB 80-158132.

19. *Security Class. (Report)*

20. *Security Class. (Page)*

21. *No. of Pages* 190

22. *Price* A09 A01

Send To:
WATER RESOURCES SCIENTIFIC INFORMATION CENTER
U.S. DEPARTMENT OF THE INTERIOR
WASHINGTON, D.C. 20240

Abstractor J. M. Harkin

Institution Wisconsin Univ. Madison

WRSIC 102 (REV. JUNE 1971)

BIBLIOGRAPHIC INFORMATION

PB80-158132

Evaluation of Mound Systems for Purification of Septic Tank Effluent.

1979

Charles J. Fitzgerald.

PERFORMER: Wisconsin Univ.-Madison. Dept. of Soil Science.
Contract DI-14-34-0001-7200

SPONSOR: Office of Water Research and Technology,
Washington, DC.
W8004112 OWRTB-099-WIS(2)
Master's thesis

See also PB80-122807.

Thirty-three mound systems for on-site disposal of septic tank effluent (STE) installed by commercial contractors in Wisconsin were surveyed and monitored for: proper design, instalation, operation, and wastewater purification. Every system examined was correctly sized and constructed but many systems were being dosed inappropriately with large volumes of STE. In these systems, bacteria and nutrients (N and P) from STE penetrate deeper into the soil than in systems being properly dosed with more frequent smaller volumes of STE. However all systems removed bacteria, viruses, phosphorus effectively and reduced the N load passing through the soil by nitrification-denitrification. Three cases of mound system failure reported in the state were due to faulty construction, not improper design or clogging. Wisconsin mound systems are expected to outlive and out perform conventional septic systems.

KEYWORDS: *Septic tanks, *Sewage treatment, *Sewage disposal, *Mound systems.

Available from the National Technical Information Service,
Springfield, Va. 22161

PRICE CODE: PC A09/MF A01

ACKNOWLEDGEMENT

Mr. Colin P. Duffy, my co-worker, assisted in all phases of the work, and as such is deserving special mention, as Mr. Duffy will also compile a Masters Thesis using selected data obtained during the project. Persons interested in the mound systems should also refer to Mr. Duffy's thesis.

The planning and execution of this research project--a joint project with Pennsylvania State University--were greatly facilitated by the advice, help, and cooperation received from personnel of the University of Wisconsin-Madison/University of Wisconsin-Extension Small Scale Waste Management Project, staff from the Bureau of the Environmental Health, Wisconsin Dept. of Health and Social Services, and the many homeowners who allowed their mound systems to be used as study objects. Special thanks are due to Patti J. Hantz for bacteriological analyses, Bruce W. Donohoe for virology analyses, Glen L. Bethel for solids and atomic absorption analyses and to William E. Enters for designing, constructing, and installing dose counters, producing field sampling equipment, and assistance in sample collection.

The project was aided financially by matching funds from the College of Agriculture and Life Sciences, University of Wisconsin-Madison, the Bureau of Environmental Health, Wisconsin Department of Health and Social Services, and the Office of Water Research and Technology, U.S. Department of the Interior, under Matching Grant Agreement No. 14-34-0001-7200.

TABLE OF CONTENTS

Acknowledgements i
Table of Contents ii
List of Figures iv
List of Tables vi

Introduction 1

Mound System Design 5

Wastewater Criteria 17

Soil Chemical and Bacterial Transformations 24
  Bacterial Attenuation 24
  Virus 26
  Nitrogen 27
  Phosphorus 30

Groundwater Quality 33
  Nitrogen 33
  Phosphorus 35

Materials 36
  Study Design 36
  Sampling Techniques 39
    General features 39
    Effluent sampling 40
    Soil sampling 41
    Groundwater sampling 45
    Chemical methods 48
    Physical properties 51
    Bacteriological determinations 51

Results and Discussion 53
  Design Criteria 53
    Adequacy of design criteria 53
      Package type 53
      Water use and retention 53
      Causes of hydraulic failure 60
  Wastewater Characteristics 64
    Nutrients 64
    Bacteria 70
    Viruses 72

Mound Performance 72
Bacterial attenuation 72
Introduction 72
All mounds 74
Package design differences 85
Individual systems 88
Factors affecting bacterial travel 94
Nitrogen 95
All mounds 95
Package design differences 102
Nitrogen transformations in individual systems 104
Phosphorus 116
All mounds 116
Individual package differences 118
Phosphorus movement in individual systems 120
Seasonal variations in mound performance 128
Clogging mat formation 131
Comparison to other systems 136
Bacterial attenuation by soil 136
Nitrate movement 145
Phosphorus movement 151
Groundwater quality 153
Prospects for Continued Satisfactory Performance 161
Recommendations 162

Literature Cited 167

LIST OF FIGURES

Figure		Page
1	Conventional septic tank/soil absorption wastewater disposal system	3
2	Typical Wisconsin mound system for on-site wastewater disposal	4
3	Various transformations of home sewage waste N in the septic tank and soil	28
4	Relationships between pH and soluble orthophosphate for various inorganic phosphates	32
5	State-wide locations of the mound systems studied	37
6	Casing and auger used to obtain soil samples within and below mounds	42
7	Cased push auger used to obtain soil samples for bacteriological analysis within and below mounds	44
8	Novel well design used for sampling groundwater near mounds	47
9	Average bacterial counts in the soil solution from the fill and natural soil below Package 1, 2 and 3 systems	80
10	Average bacterial counts in the fill of and natural soil under systems 1-40, 3-34 and 3-64	81
11	Average bacterial counts in the fill of and natural soil under systems 1-160 and 3-61	82
12	Average bacterial counts in the fill of and natural soil under systems 3-47, 3-69, 3-71 and 3-82	83
13	Average bacterial counts in the fill of and natural soil under systems 1-161, 2-11, 3-78 and 3-83	84
14	Average TKN, NH_4-N, NO_3-N and Cl^- in the fill of and natural soil under 33 mounds	97

Figure Page

15 Average TKN, NH_4-N, NO_3-N and Cl^- in the fill of and natural soil below Package 1, 2 and 3 mounds 103

16 Average TKN, NH_4-N, NO_3-N and Cl^- in the fill of and natural soil under systems 1-40, 3-34 and 3-64 106

17 Average TKN, NH_4-N, NO_3-N and Cl^- in the fill of and natural soil under systems 1-160 and 3-61 108

18 Average TKN, NH_4-N, NO_3-N and Cl^- in the fill of and natural soil under systems 3-82, 3-69, 3-71 and 3-47 .. 110

19 Average TKN, NH_4-N, NO_3-N and Cl^- in the fill of and natural soil under systems 1-161, 2-11, 3-78 and 3-83 . 113

20 Average TP and OP in the fill of and natural soil under 33 mound systems 117

21 Average TP and OP in the fill of and natural soil below Package 1, 2 and 3 mounds 119

22 Average TP and OP in the fill of and natural soil under systems 3-34 and 3-64 121

23 Average TP and OP in the fill of and natural soil under systems 1-160 and 3-61 123

24 Average TP and OP in the fill of and natural soil under systems 3-47, 3-69, 3-71 and 3-82 124

25 Average TP and OP in the fill of and natural soil under systems 1-161, 2-11, 3-78 and 3-83 126

26 Average TOC, TKN and TP in the fill of and soil under all systems sampled from Oct. 1977 to Aug. 1978 130

27 TKN and TOC at the gravel/sand interface and in the fill of and natural soil under systems 3-47, 3-72, 3-78 and 3-82 134

28 Location of groundwater wells around systems 3-47, 3-72, 3-78 and 3-82 154

LIST OF TABLES

Table Page

1 Site and design criteria for the Wisconsin mound systems 10

2 Particle size distribution of some Wisconsin sands and their suitability for use as a fill in mounds 12

3 Average daily water use found in several studies 18

4 Some chemical and physical parameters of raw wastewater and septic tank effluent 19

5 Data for bacteria, pH, and temperature on effluents from five septic tanks 23

6 Water use characteristics for 20 Wisconsin mound systems .. 55

7 Estimated flow rate through a medium sand (cm/day) for various dosing rates assuming an infiltration time of 1 hour for each dose volume 57

8 Temperature in °C of effluent in pump chambers 63

9 Average wastewater characteristics for septic tank effluent taken from 33 mound systems 65

10 Bacterial attenuation ratios, soil texture and moisture content for the fill and at various depths below selected mound systems with fill characteristics, hydrologic loading and soils data 75-76

11 Bacterial attenuation ratios and moisture contents in the fill of and at increasing depths below mounds of Package 1, 2 and 3 designs 77

12 Bacterial attenuation ratios and moisture contents for the fill of and soil at increasing depths below all systems studied 77

13 Bacterial attenuation ratios for the dry soil in the fill of and at increasing depths below Package 1, 2, and 3 mounds, and averages for 33 mounds 78

Table		Page
14	Bacterial attenuation ratios for the soil solution in the fill of and at increasing depths below Package 1, 2 and 3 mounds, and averages for 33 mounds	79
15	Bacterial counts in surface soils near mounds calculated on a soil solution basis	86
16	Hydraulic loading and fill characteristics of selected mounds	90
17	TKN, NH_4-N and NO_3-N in surface soils from areas surrounding several mounds	99
18	TP and OP for surface soils from areas surrounding several mounds	129
19	Total carbon:total nitrogen ratios for soils in and below four mound systems	135
20	Bacterial counts obtained below 2 conventional wastewater disposal systems in coarse-textured soils (Plainfield sand)	144
21	Results of bacterial and chemical analysis of water samples from the pumping chambers of and groundwater around systems 3-47, 3-72, 3-78 and 3-82	156-157

INTRODUCTION

In 1970, the United States census determined that there were 371,567 septic systems operating within the State of Wisconsin. From 1970 to 1978, 182,598 additional septic tank permits were issued; of these only 13% were for replacement septic systems (Miller, 1978). The number of septic tank permits issued each year has increased from 14,000 in 1970 to 23,000 in 1978. Thus there are probably now over 500,000 septic systems operating in the State of Wisconsin.

Assuming an average family size of 3 persons, approximately 1.5 million (or one-third) of the 4.6 million population of the State of Wisconsin therefore now rely upon on-site wastewater disposal systems, and the number is increasing by about 75,000 persons per year (Miller, 1978). However, over 50-60% of the soil in Wisconsin is unsuitable for accepting waste waters due to reasons of intrinsic impermeability, high or perched water tables, seasonal inundation, shallow soil depth to bedrock, or excessive slope (U.S. Geol. Survey, 1976a).

Section H62.20 of the Administrative Code of the State of Wisconsin presently restricts installation of conventional septic tank effluent subsurface soil absorption systems to soils with percolation rates of 60 min/in or faster and with a depth to groundwater or bedrock greater than 5 feet. The percolation rate is the time it takes for 1 inch of water to be absorbed from a prewetted test hole at a depth where the absorption system is to be placed (C. H62.20 Wisconsin Administrative Code as in effect August 1, 1976). The mound systems to be discussed are designed to allow for installation of elevated soil absorption systems in areas with soils having percolation rates of up to 120 min/in and depth to groundwater or bedrock of as little as 2 feet (Converse, 1978).

The conventional septic sysem (Fig. 1) comprises a septic tank and a soil absorption area; both components may have various designs. The septic tank normally consists of a large chamber usually made of steel, concrete, or fiberglass, with an entry and exit pipe. Baffles or sanitary tees are installed to restrict surge flow and to prevent floating material (scum) from entering the soil absorption area or clogging the house sewer. In the septic tank the wastewater undergoes anaerobic bacterial digestion and settling of suspended material to form a sludge. The wastewater or effluent leaving the septic tank enters a 4-inch diameter pipe and flows to the soil absorption area. Here the effluent trickles through a 4-inch diameter perforated pipe and into a gravel bed for further treatment by passage through the subsoil.

A typical Wisconsin mound system (Fig. 2) consists of a regular septic tank followed by a pumping chamber which collects the effluent. A submersible pump introduces the effluent in intermittent doses under pressure through a distribution network into an absorption bed or trenches installed in a suitable fill material which is superimposed over the natural soil surface and which is usually more permeable than the subsoil. The mound designs developed by the University of Wisconsin are modifications of the NoDak mound systems (Witz, 1975) and attempt to overcome some of the restriction which prohibit the installation of a conventional subsoil absorption system in problem soils.

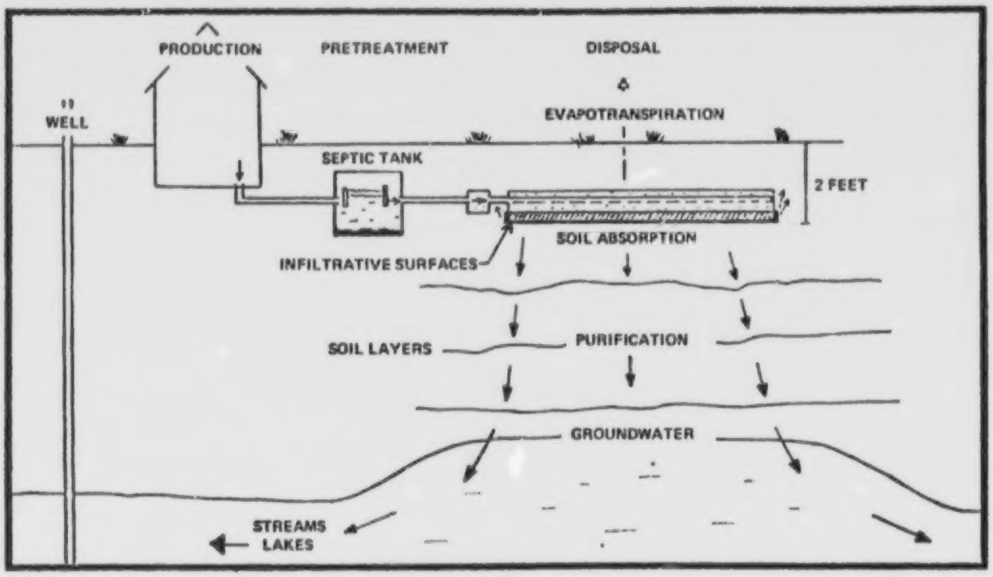
PRODUCTION
PRETREATMENT
DISPOSAL
EVAPOTRANSPIRATION
WELL
SEPTIC TANK
2 FEET
SOIL ABSORPTION
INFILTRATIVE SURFACES
SOIL LAYERS
PURIFICATION
GROUNDWATER
STREAMS
LAKES

Fig. 1. Conventional septic tank/soil absorption wastewater disposal system.

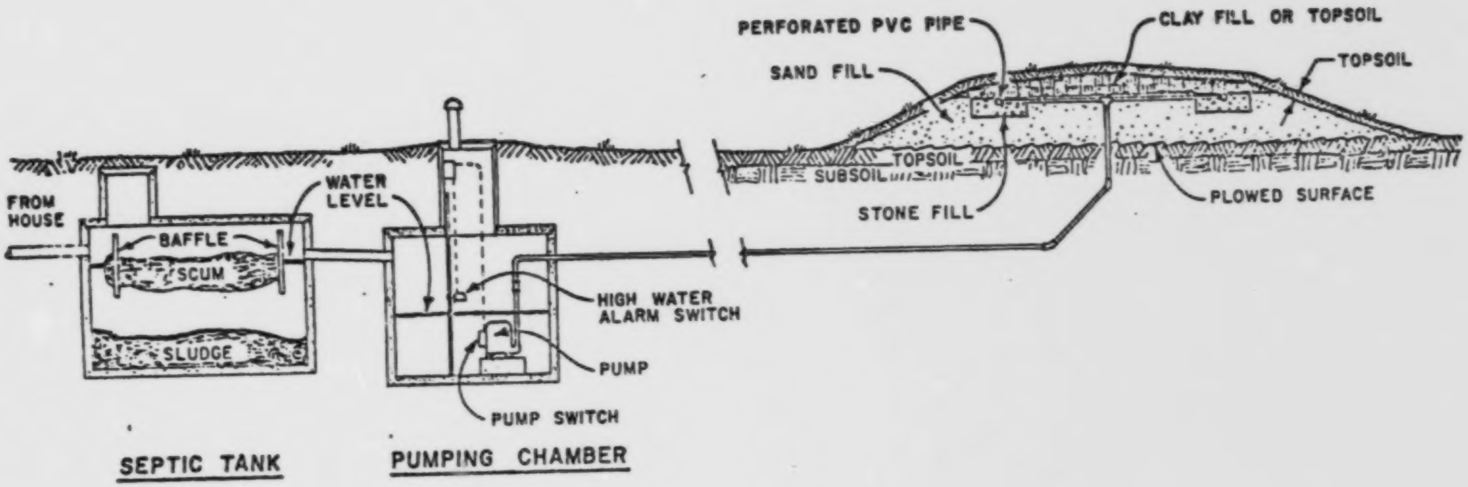
PERFORATED PVC PIPE
CLAY FILL OR TOPSOIL
SAND FILL
TOPSOIL
TOPSOIL
SUBSOIL
STONE FILL
PLOWED SURFACE
FROM HOUSE
WATER LEVEL
BAFFLE
SCUM
SLUDGE
HIGH WATER ALARM SWITCH
PUMP
PUMP SWITCH
SEPTIC TANK
PUMPING CHAMBER

Fig. 2. Typical Wisconsin mound system (Package 1) for on-site wastewater disposal.

The objective of this project was to study mound wastewater disposal systems constructed by commercial installers in the State of Wisconsin and to evaluate the design, construction, installation and maintenance of the systems in terms of their efficiency for satisfactory long-term wastewater purification in problem soil areas.

Mound System Design

The Wisconsin mound design (Fig. 2) attempts to overcome some of the soil limitation problems that restrict the installation of a conventional subsurface soil absorption system (Fig. 1), viz. shallow soil depth or intrinsic impermeability. The mound design does this by placing an overburden of medium sand on top of the natural soil surface to obtain a minimum of 3 feet of soil and fill for the treatment of the septic tank effluent. The Wisconsin mound designs are divided into 3 types depending upon the soil and site limitations. These 3 mound designs are for:

(1) Slowly permeable soils with or without high groundwater;
(2) Shallow permeable soils over pervious or creviced bedrock;
(3) Permeable soils with high water tables.

The 3 mound designs are summarized in packages which give step-by-step procedures for the design and construction of a mound system (Converse et al., 1976a,b,c; Converse, 1978); these are available from the Small Scale Waste Management Project, 1 Agricultural Hall, University of Wisconsin, Madison, Wis. 53706. The site criteria for the installation of the Wisconsin mound designs are summarized in Table 1.

Mound systems have 3 features which differentiate them from the normal septic tank subsurface disposal system: the pumping chamber, the distribution network, and the mound itself.

The pumping chamber collects the septic tank effluent until a predetermined volume is obtained, then a submersible pump or siphon introduces the bulk of the collected effluent in a single dose into the pressurized distribution system. The volume of septic tank effluent dispersed with each dosing event is usually controlled by float switches located in the pumping chamber to give a specified dose volume.

The pressurized distribution network is a lattice of 1-1½ inch PVC pipe perforated with small, widely spaced holes which is laid in the gravel absorption bed of the mound. The purpose of the pressurized distribution network is to introduce the septic tank effluent into the gravel from the pumping chamber with an even distribution over the total area of the absorption bed and thus prevent the localized overloading that occurs in a traditional gravity feed system (Otis et al., 1974). Localized overloading leads to the formation of a crust or clogging mat (Laak, 1970, 1976; Harkin et al., 1976; Jawson, 1976) that often causes premature failure of conventional systems. In addition, proper use of the pressurized dosing system allows for the maintenance of slow unsaturated flow through the finer pores in the soil within the fill material and thus promotes bacterial and chemical purification of the water (Bouma et al., 1972; Bouma, 1975).

Soil clogging or crust formation is the most common, perhaps the only natural, mechanism that interferes with the proper operation of a properly designed and correctly installed and maintained septic tank-soil absorption system (Harkin et al., 1976). The effluent leaving the septic tank and flowing to the soil absorption field contains various amounts of dissolved and suspended material, both organic and inorganic, and high numbers of bacteria. As the effluent trickles through the gravel and enters the soil, the fine texture of the soil removes and treats these materials by physical, chemical, and biological processes. In time a crust or layer forms which often has a markedly lower hydraulic conductivity than the natural soil at the gravel-soil interface. The reduction in hydraulic conductivity, the amount of water that percolates through the soil in a given period of time, is due to the filling of the soil pores with deposits of inorganic precipitates, organic residues, and bacteria that feed on the organic material (Laak, 1970, 1976; Harkin et al., 1976). This biological crust impedes the movement of water and creates a zone of unsaturation in the natural soil below the crust (i.e. all the available pores in the soil below the crust are not conducting water). In many normal septic systems, water is ponded above such a crust; the rate of the water movement through the crust, though not fast enough to fill all the soil pores below the crust, is still fast enough to allow the system to dispose of the total water input without surface discharge of untreated effluent. In such situations, the biological crust enhances the purification of the septic tank effluent by improving the filtration properties of the soil and providing a substrate or energy/food source for the chemical and biological processes that remove contaminants from the water.

Thus, soil crusting actually aids in wastewater treatment provided the crust does not impede percolation to an extent where input exceeds outflow and surface seepage results. In a funtioning ponded system, the crust establishes an equilibrium which balances the input-output seepage flow and which is governed by the unique environment within each particular system. This environment is governed by factors such as the effluent quality (solids content, biological makeup, pH, etc.), loading rate and dosing schedule, soil or filter medium texture, and the microbial populations and activities. It is yet unknown exactly what factors or combination of factors cause such a drastic change in the permeability of the crusted layer that the inflow-seepage balance in the ponding situation is altered and surface discharge results. It is suspected that abrupt changes in the microbial environment may be responsible (Jawson, 1976). The crusted condition can be reversed by treatment of the soil with hydrogen peroxide (Harkin et al., 1976) according to the POROX® process (Harkin, 1977).

In the mound portion of the system, a soil with properties known to favor acceptance and purification of wastewater is mounded on top of the original soil at the installation site to supplement the acceptance of septic tank effluent into slowly permeable soils or the purification of effluent in shallow or porous soils over high water tables or fissured bedrock. Within the mounded soil is placed a conventional gravel-filled bed or trench system into which the septic tank effluent is introduced through the distribution network; widely spaced narrow trenches are preferred at sites with slowly permeable soils to ensure that an adequate area is available for water to soak evenly into the ground and to prevent intrusion of mounded water tables into the system (Bouma, 1975). The amount of "fill" material added above the native soil is designed to ensure that a sufficient depth, usually a minimum of 3 feet, of good soil and fill is available to purify the effluent before seepage reaches the groundwater: thus 1 foot of fill is usually enough between the bottom of the gravel and the natural soil surface over slowly permeable soils or permeable soils where the water table never encroaches above 2 feet from the surface, but 2 feet of fill are needed below the gravel over shallow permeable soils over high water tables or pervious bedrock (Table 1). The mound is covered with a cap of clayey soil to promote runoff rather than infiltration of precipitation and to provide protection from frost. Good quality topsoil is then added to encourage growth of grass or other ground cover to prevent erosion of the mound and enhance runoff and frost protection.

The fill material and its placement are the most costly and important aspects of mound construction (Converse, 1978). A medium sand in conjunction with the dosing network can be used to apply the effluent to the mound in such a way that the flow is in either an unsaturated or nearly saturated state. The type of flow through the mound governs the level of bacterial removal and chemical treatment of the septic tank effluent, with optimum performance (purification at acceptable flow rates) occurring with slow, unsaturated flow through a medium sand (Bouma et al., 1972).

In laboratory soil columns rates of removal of bacteria from percolating septic tank effluent are high under conditions of unsaturated flow (Ziebell et al., 1975) but much less under saturated flow (McCoy and Ziebell, 1975). Removal of chemical impurities from septic tank effluent is also improved by aerobic conditions associated with unsaturated flow over the anaerobic state usually associated with saturated conditions (Bouma et al., 1972).

A medium sand with an unsaturated flow regime is the best

Table 1. Site and design criteria for the Wisconsin mound systems (Converse, 1978)

Restricting factors	Soil group Slowly permeable soils (Package 1)	Permeable soils with pervious bedrock (Package 2)	Permeable soils with high water tables (Package 3)
Percolation rate[a]	60–120 min/in	3–60 min/in	3–60 min/in
Depth to pervious rock	24 in	24 in	24 in
Depth to high water tables	24 in	24 in	24 in
Depth to impermeable soil layer or rock strata	60 in	60 in	60 in
Slope	6%	12%[b]	12%[b]
Fill depth	1 ft	2 ft	1 ft
Absorption area configuration	Trench	Bed	Bed

[a] Percolation test depth is 24, 12, and 24 inches for slowly permeable, shallow, and high water-table soils, respectively, unless there is a more restrictive horizon above. If perched water is at 24 inches, the test depth should be 16 inches.

[b] For a percolation rate of 3–29 min/in, the maximum slope is 12%; for a percolation rate of 30–60 min/in, the maximum slope is 6%.

combination for wastewater renovation in the fill in a mound system (Bouma, 1975; Magdoff et al., 1974). The recommended sand fill for a mound system is a medium sand (Converse et al., 1976a-c; Converse, 1978); this includes 25% or more of coarse, very coarse, or medium sand (>0.25 mm), less than 50% fine or very fine sands (<0.25 mm), and not more than 10-15% fines or very fine sands (<0.05 mm) (USDA Classification). The nature of the fill material is of prime importance in dictating the purification properties of a mound system: fill material that is too coarse reduces the filtration properties of the fill; fill material that is too fine promotes near saturated flow and anaerobic conditions that retard the breakdown of organic matter. Table 2 shows the particle size distribution of some Wisconsin sands and their suitability for use as a fill in the mound designs (Converse, 1978). Assuming that the fill placed in a mound is the medium sand recommended, the efficiency of wastewater purification then depends upon the rate of flow of the water through the fill. The flow rate is determined by whether the flow is saturated or quasi-saturated (all or nearly all of the soil pores are filled with and are conducting liquid) or unsaturated (only the finer soil pores are filled with and are conducting water). Under saturated or quasi-saturated conditions, indicated by high soil moisture contents, flow is fast and purification is slight; under unsaturated conditions, indicated by low soil moisture contents, flow is slow and purification is high. Whether the flow through a medium sand fill material is saturated or unsaturated depends upon the size of the absorption area, the hydraulic characteristics of the dosing system, and the presence or absence of a biological crust at the interface between

Table 2. Particle size distribution of some Wisconsin sands and their suitability for use as a fill in mounds (after Converse, 1978)

	USDA Classification								
	Gravel								
Sample no.	Gravel >2 mm	Very coarse sand 1-2 mm	Coarse sand 0.5-1 mm	Medium sand 0.25-0.5 mm	Fine sand 0.1-0.25 mm	Very fine sand 0.05-0.1 mm	Silt and clay <0.05 mm	Unif. Coeff. Approx.	Comments
	%	%	%	%	%	%	%		
1	9	6	22	51	11	0	1	2	Good fill
2	5	0	27	51	13	1	1	2	Good fill
3	0	16	16	39	34	4	1	2	Good fill
4	46	13	12	14	6	2	7	10	Unacceptable - too coarse
5	40	15	17	18	7	1	2	4	Unacceptable - too fine
6	0	0	7	33	55	4	1	a	Unacceptable - too fine
7	2	0	1	4	30	10	53	a	Unacceptable - too fine
8	18	3	8	30	24	4	15	10	Poor to fair - judgement[b]
9	23	15	22	20	10	6	4	10	Poor to fair - judgement[b]

[a] Sample is not a medium sand.

[b] Sample 8 has too many fines. Sample 9 is too coarse. Both of these would be judgement decisions. If acceptable fills were near, these two fills would not be recommended.

the gravel in the absorption bed and the sand fill. To plan suitable hydraulic loading rates for a mound system, both the crusted and uncrusted situations must be considered because the pressurized dosing network helps to retard or prevent crust formation in mound systems by applying the principle of dosing and resting (Bouma et al., 1974a), although the crusted condition may eventually develop.

The saturated hydraulic conductivity of an uncrusted medium sand, i.e. the amount of water that will flow through the sand under unit hydraulic head (1 cm), was determined in field measurements to be approximately 480 cm/day, i.e. water flowing through medium sand would soak in at a rate of 115.2 gal/day/ft^2 if all of the soil pores were conducting water (Bouma et al., 1972). Quasi-saturated flow (flow near the saturated level) allows for little bacterial attenuation (Bouma et al., 1972). The maximum levels of bacterial removal from septic tank effluent flowing through uncrusted sands are accomplished with flows not exceeding 40-50 cm/day (9.6-12.0 gal/day/ft^2 if the water were seeping in continuously all day).

The amount of liquid applied to the mound in practice is expressed in terms of a daily loading rate, the number of gallons of septic tank effluent applied per day to each square foot of bottom area of the absorption field. To ensure adequate purification of septic tank effluent, the absorption area in a mound should therefore be sized large enough to keep the flow through the uncrusted fill at least below the level of 40-50 cm/day even when the water is applied in doses by the pump, and not as a continuous trickling flow under gravity, as in a conventional system.. Consequently, in practice, much larger absorption areas and therefore much lower loading rates are planned into the design, in case an organic clogging mat or crust should form over an extended period of time at the gravel/sand fill interface (Bouma et al., 1972).

Crust test procedures for measuring hydraulic conductivity (Bouma et al., 1972, 1974b) indicate that the maximum flow through a crusted medium sand is only 5 cm/day (1.2 gal/day/ft^2). This is therefore the maximum loading rate used in practice to size the absorption field in mound systems.

To size an absorption field for a mound system, the total daily wastewater load expected is calculated, assuming a water usage of 150 gals/bedroom/day in homes or corresponding volumes for other types of buildings (Converse et al., 1976a-c; Converse, 1978). Thus, for a 3-bedroom home, a daily wastewater load of 450 gallons is used. This volume is divided by the loading rate of 1.2 gal/day/ft^2 to derive the bottom area in square feet of the gravel absorption bed or trenches in the sand fill (450 x 1.2 = 375 ft^2 for a 3-bedroom home).

In order to keep the instantaneous flow rate through uncrusted fill immediately following dosing below the safe value of 40-50 cm/day, it is recommended in the design packages (Converse et al., 1976a-c; Converse, 1978) that the septic tank effluent be applied to the mound in 4 doses per day. This mode of application brings extra benefits. The rates of breakdown of organic matter and loss of fixed nitrogen by denitrification in soil are greatest when a system cycles from anaerobic to aerobic conditions every 6-12 hours (Patrick and Reddy, 1974, 1976). A mound receiving 4 doses with anaerobic septic tank effluent every day would have 6-hour periods of unsaturated flow between doses. With unsaturated flow, the sand fill would retain an aerobic state, which would promote bacterial die-off and accelerate breakdown of organic matter (Alexander, 1977).

Thus, for a 3-bedroom home, producing 450 gallons of septic tank effluent, a volume of $450 \div 4 = 115$ gallons of liquid should be applied with every dose to the 375 ft^2 of absorption area. The dosing volume of 115 gallons is achieved by setting limit switches (usually float switches) at appropriate levels in the pumping chamber.

It is useful at this point to introduce two further concepts which help describe the hydraulic flow through mounds but which have not traditionally been used in mound installation guideline packages. These are the dosing rate, or volume of liquid applied per square foot of absorption area per dose, and the dosing frequency, or number of doses per day. For a 3-bedroom home constructed and operated according to the engineering package instructions, the dosing frequency is 4 doses per day, and the dosing rate is 115 gallons $\div$ 375 ft^2 = 0.3 gal/ft^2/dose.

These new terms are useful in helping to describe the flow in mound systems which are not being loaded daily in multiple applications. For example, a mound system serving a recreational home receives only intermittent use, so that regular doses of septic tank effluent are not applied daily to the mound. The dosing rate therefore becomes more important for describing the loading of the mound and flow through the fill while the system is actually in use. The dosing rate, multiplied by the dosing frequency, is the loading rate for the periods of active use of the mound. The concept of dosing frequency can only be aptly applied to systems where several doses per day are pumped into the mound while the mound system is being used. Where systems are not in constant use, for example, at weekend or recreational homes, the mound may receive a dose of septic tank effluent, for instance, only every 3 weeks. It is better is such cases to note the number of days between doses rather than the dose frequency (doses/day).

The combination of a pressurized distribution system, a medium sand fill, a properly sized absorption field, a properly adjusted dosing volume, and a dosing frequency of 4 doses/day should guarantee that a mound system serving any household with a per capita water use of up to 75 gal/person/day (or 150 gal/bedroom/day) should always have unsaturated flow through the fill and into the underlying soil, assuring good septic tank effluent purification regardless of whether the sand fill is crusted or not. Such a system is expected to operate successfully for a practically unlimited period of time. The mound monitoring project described in the following was designed to examine whether these expectations are achieved in practice in systems built by commercial installers in Wisconsin.

WASTEWATER CRITERIA

Wastes produced in a building served by an on-site disposal system flow from the sewer into a septic tank. The septic tank (Figure 1 and 2) is a water-tight container where the wastewater undergoes primary clarification and anaerobic bacterial digestion.

Water use within the home or building being served dictates the character of the raw effluent entering the tank. Witt et al. (1974), Bennett et al. (1974), and Siegrist et al. (1976) studied the characteristics of wastewater from rural homes and determined the inputs to a septic tank. Table 3 reproduced from Witt et al. (1975) indicates the total wastewater flow arising from household water use.

In the septic tank a majority of the functioning or "active" bacteria are facultative anaerobes, i.e. they grow under both aerobic (soil absorption field) and anaerobic (septic tank) environments. The function of the flora of a septic tank is to perform the hydrolysis and dissimilation of a variety of a variety of complex compounds such as proteins, carbohydrates, and fats.

Table 4 (from Ziebell et al., 1975) indicates the results of analysis of samples from three household wastewater disposal systems, giving the values of some of the chemical components of raw wastewater and septic tank effluent. Comparison of the percentage of NH_4-N in the raw wastewater (22%) and in the septic tank effluent (70%) indicates that considerable deamination is occurring in the septic tank. Similarly, the hydrolysis of proteins is indicated by

Table 3. Average daily water use found in several studies.

	USGS 1962	Haney & Hamann 1967	Laak 1971	Ligman 1972	Wallman 1972	Ontario Research 1973	Bennett 1973	Witt 1974
Toilet	41	45	43	41	27-45	38	33	22
Laundry	4	5	16	19	18	12	26	25
Bath	37	30	19	26	18-36	34	20	23
Kitchen	6	6	8	10	13	10	12	11
Cleaning	3	4	5	1	---	3	3	--
Drinking	5	3	3	3	---	3	3	--
Miscellaneous	4	7	6	0	6	0	3	13[1] 6[2]
Flow (gpcd)	--	--	41	45	30-50	--	44	43

Source: Witt (1974).

[1] Other.
[2] Water softener.

Table 4. Some chemical and physical parameters of raw wastewater and septic tank effluent (Ziebell et al., 1975).

Parameter (mg/L)	Raw wastewater		Septic tank effluent	
	Mean	Relative %	Mean	Relative %
BOD_5, unfiltered	343		158	
BOD_5, filtered	212		120	
Suspended solids				
total	259		50.8	
volatile	203	78	35.5	70
Ammonia-N	8.8	22	38.7	70
Organic N	30.3	74	16.0	29
$NO_2 + NO_3$-N	1.8	4	0.6	1
Ortho-P	9.4	45	11.5	79
Organic-P	11.3	55	3.1	21

the relative percentages of organic nitrogen and phosphorus (74% organic N and 55% organic P in the raw wastewater versus 29% and 21% in the septic tank effluent). These changes are due to the enzymatic activity of the septic tank flora. In addition, approximately 80% reduction in suspended solids can be attributed to hydrolysis by bacteria or settling of the solids (Baumann et al., 1977). During the bacterial digestion (biodegradation) of the wastes, the products of hydrolysis are used for growth and respiration (energy) and account for the reduction of the BOD from the raw effluent (343 mg/L) to the septic tank effluent (158 mg/L); however, this still leaves a relatively high BOD to be degraded in the soil absorption field. Other studies have indicated similar treatment levels of raw effluent in the septic tank (Bouma et al., 1972).

These results indicate the anaerobic nature of the biological digestion in septic tanks by the low NO_3-N content and predominance of NH_4-N as the major form of nitrogen in septic tank effluent. One study suggests that an average family of four will contribute approximately 33 kg N/year to a soil absorption field (Walker et al., 1973).

The predominant sources of phosphorus in septic tank effluent are detergents with phosphorus builders (Sawyer, 1965) and human excreta (Sherman, 1952). Other studies have indicated that the total contribution of phosphorus to a septic tank results in a load of 0.8 kg of P/person/year (Otis et al., 1975) to 3.0 kg of P/person/year (Bernhard, 1975).

In the septic tank about 85% of the phosphorus is converted to soluble orthophosphate (Magdoff et al., 1974; Otis et al., 1975). IES (1979) summarized published data and determined that the average phosphorus in septic tank effluent is 23.5 mg/L of total P and 16.37 mg/L of ortho-P; thus, 70% of the total phosphorus is in the soluble ortho-P form.

Bennett et al. (1974) found the average total solids content of raw wastewater to be approximately 1180 mg/L. A septic tank removes solids primarily by settling and flotation. Approximately 30-40% of the floating and settled solids are decomposed by anaerobic bacterial action (Bennett et al., 1974). The resultant sludge accumulates in the bottom of a septic tank or floats as a scum on the surface of the tank.

Baffles and/or sanitary tees are installed in a septic tank to reduce surge flows, increase transit time for the maximum removal of solids, and prevent fouling of the inlet and outlet pipes by scum or solids. Baumann et al. (1977) determined that a well functioning septic tank will retain 98-99% of the "settleable" solids and 60-90% of the suspended solids dependent upon the retention time of the effluent in the septic tank (27-34 hours). Ziebell et al. (1975) concluded that an 80% reduction can be expected in suspended solids in a septic tank.

Laak (1976) concluded that the crust or clogging mat formation in the soil absorption field is proportional to the five day BOD and total suspended solids found in septic tank effluent:

$$\text{Clogging} \propto \left[0.04\ (BOD_5 + TSS)\right]^{1/3}$$

modification from Laak (1976).

The bacterial population in a septic tank is primarily derived from enteric bacteria from the human digestive tract, high numbers of coliform bacteria including fecal coliforms, fecal streptococci, and Pseudomonas aeruginosa. These organisms are traditionally used as indicators of sewage contamination in wastewater.

Ziebell et al. (1975) reported levels of indicator organisms in septic tank effluent (Table 5). These counts and a 59% occurrence of salmonellae in septic tank effluent indicate that septic tank effluent is a potential source of pathogens and is unsafe for human contact.

Viruses from the gut of an infected host using a septic system enter the septic tank and may sorb onto and settle out with larger solids. The virus is too small to settle out by itself so its removal in the septic tank is thought to follow closely the removal of solids by settling (Cliver, 1975); an 80% reduction of virus in the septic tank can be expected, leaving 20% to enter the soil absorption field.

Table 5. Data for bacteria, pH, and temperature on effluents from five septic tanks (Ziebell et al., 1975).

Bacteria[1]	Mean	95% confidence interval of mean	Range
Fecal streptococci per 100 mls	3,800(97)[2]	2000-7200	<100-1,000,000
Fecal coliforms per 100 mls	420,000(94)	290,000-620,000	500-18,000,000
Total coliforms per 100 mls	3,400,000(91)	2,600,000-4,400,000	150,000-40,000,000
Ps. aeruginosa per 100 mls	10,000(13)	1,900-54,000	210-350,000
Total bacteria x 10^5 per ml	34(58)	25-48	0.3-2300
pH	7.3(58)	7.2-7.4	6.4-8.0
Temperature in tank, °C	17(13)	15-19	12-23

[1] log normalized data

[2] number of samples used in calculation

SOIL CHEMICAL AND BACTERIAL TRANSFORMATIONS

Bacterial attenuation

Bacterial attenuation, or the removal of bacteria from percolating effluent by the soil, is primarily governed by the soil texture and water movement (saturated or unsaturated flow). The soil texture is important because the soil serves as a filter medium to physically entrap bacteria. The mode of water movement is important as it has been shown that bacterial travel in an unsaturated condition is minimal (Bouma, 1973) whereas bacterial travel in a saturated condition can be extensive (McCoy and Ziebell, 1975). Under saturated flow conditions bacterial attenuation depends upon their survival time and the distance of soil moisture movement in that time (McNabb et al., 1977).

Diseases such as hepatitis (viral), typhoid, cholera, salmonellosis, and gastroenteritis have all been traced to groundwater contamination by subsurface wastewater disposal systems (McCoy and Ziebell, 1975). The USGS (1979) has attributed high coliform counts in Door County, Wisconsin groundwater to contamination by septic systems as a result of shallow soils over creviced bedrock which allow bacterial travel into the aquifer.

Investigations for evidence of human fecal contamination entails tests for indicator organisms that are continually present in human excreta. Ziebell et al. (1975) examined four fecal organisms in septic tank effluent, viz. total and fecal coliforms, fecal streptococci and *Pseudomonas aeruginosa*. Tests for total and fecal coliforms have long been established and are considered reliable indicators of human fecal contamination (Kott, 1976). However, the test for fecal streptococci or enterococci (Standard Methods, 14th Ed., 1975) has come under criticism as it may give positive results due to streptococci from animal, vegetable, and insect origins. In addition, the possibility of the fecal streptococci survival for extended periods in soil has been suggested as a problem in indicating recent human fecal contamination (Clausen et al., 1976).

Tests for *Pseudomonas aeruginosa* are in the development stage: it has been suggested (Kott, 1976) that this is a possible "new" indicator bacteria of human fecal contamination in surface waters. Problems have developed in this theory, especially in soils, as it has been indicated that *Pseudomonas aeruginosa* is a voracious heterotroph capable of growth outside the human gut (Hoadley, 1976). Ziebell et al. (1975) also observed growth of *Pseudomonas aeruginosa* in columns representing mound disposal systems; more *Pseudomonas aeruginosa* emerged from columns than were present in the effluent being introduced. Testing for *Pseudomonas aeruginosa* will undoubtedly continue as it is an opportunistic pathogen of public health concern, but its ability to indicate recent fecal contamination is questionable.

Virus

Green and Cliver (1975) studied the movement of virus in soil columns and found that "a properly operating sand filtration system should produce effluents that present no hazards from human enteric viruses." However, temperature, dose rate, and "conditioning" had a marked effect upon virus travel through sand columns. Virus removal was lower in sands at 3°C than in sands at room temperature (37 versus 97% reduction of virus numbers). It was postulated that a major factor in virus removal was bacterial digestion of virus; thus, if microbial activity is lowered by temperature, so is viral removal. In addition, the dose rate affected viral travel in much the same way as bacterial travel: a very high dose rate (50 cm/day) resulted in less virus removal than a low dose rate (5 cm/day) which removed all virus from the effluent.

Surprisingly, a conditioned sand column (i.e. one that had already received septic tank effluent) showed less virus removal than a clean sand column. Virus retention by the fill decreased markedly after a few weeks of operation (Green and Cliver, 1975). An overloaded operating soil absorption field may therefore not perform effective virus removal.

Nitrogen

Nitrogen transformations below a soil absorption field are depicted in Figure 3. The major forms of nitrogen in septic tank effluent are NH_4-N and organic nitrogen with a carbon to nitrogen ratio of 10 (Sikora and Corey, 1976). In time, the organic nitrogen in septic tank effluent should also be mineralized to NH_4-N. The carbon in septic tank effluent arises from bacterial mass and high carbon wastes such as toilet paper, food wastes, etc. and is not associated with the nitrogen which is primarily in the NH_4-N form.

NH_4-N entering a soil absorption field will be sorbed almost immediately by cation exchange onto soil and organic matter particles. Because a rapid equilibrium is set up between solution and sorbed NH_4-N, ammonium is in an available form (Sikora and Corey, 1976). The microbial population, given an available carbon source, will utilize solution NH_4-N and immobilize it into microbial tissue with a carbon to nitrogen ratio of approximately 10. This organic nitrogen is in a form relatively resistant to further degradation (Bartholomew, 1965). Bremner (1965) estimated that approximately 1-3% of the total nitrogen in soil organic matter becomes nonbiodegradable. Consequently a similar fraction should remain part of the organic crust in a soil absorption field.

Immobilization of inorganic combined nitrogen forms (NH_4-N and NO_3-N) in soil occurs under both aerobic and anaerobic (unsaturated and saturated) conditions but, as with many biological

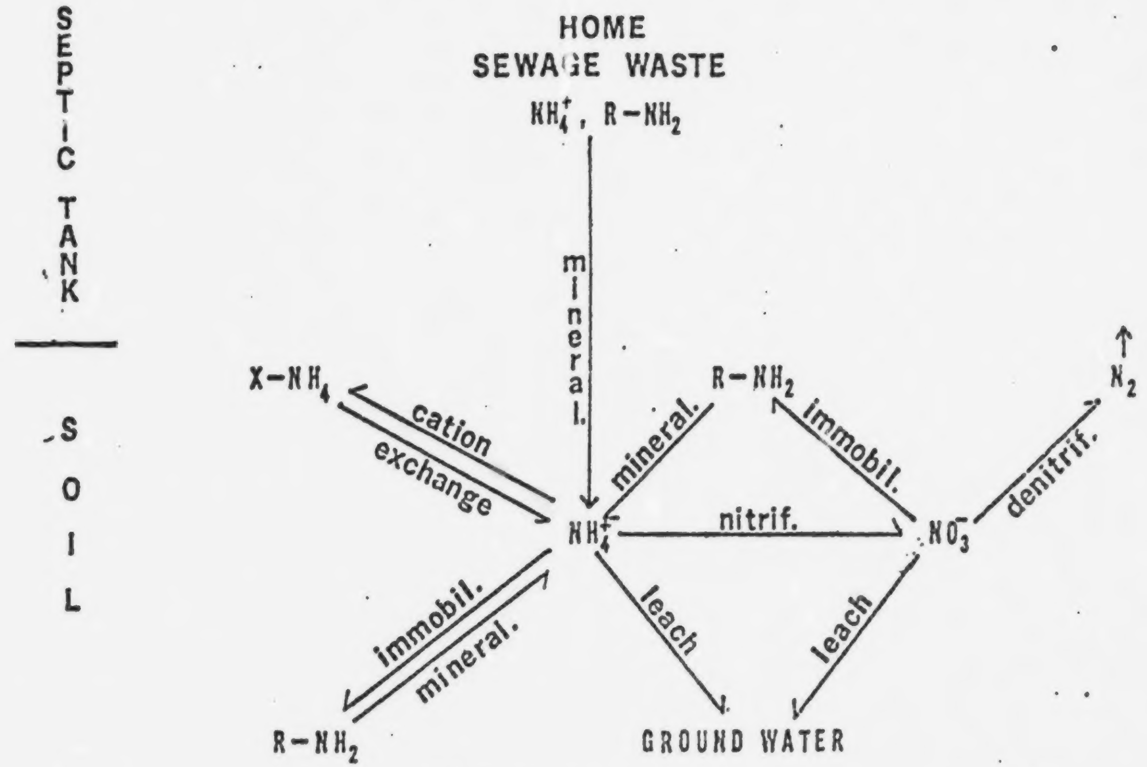
SEPTIC TANK
SOIL
HOME SEWAGE WASTE
NH_4^+, $R-NH_2$
mineral.
$X-NH_4$
cation exchange
NH_4^+
mineral.
$R-NH_2$
immobil.
nitrif.
NO_3^-
denitrif.
N_2
immobil.
mineral.
$R-NH_2$
leach
leach
GROUND WATER

Figure 6 Various transformations of home sewage waste N in the septic tank and soil.

from Sikora and Corey (1976).

reactions, occurs more rapidly under aerobic conditions (Alexander, 1977). Under anaerobic conditions NH_4-N can be leached into groundwater. As the cation exchange sites in a soil absorption field become saturated with the various cations present in septic tank effluent, immobilization may be slow enough to allow NH_4-N movement into groundwater (Sikora and Corey, 1976). Such movement was observed by Magdoff et al. (1974), who surmized that the major factors affecting NH_4-N movement into groundwaters under anaerobic conditions would be the number of cation exchange sites exposed to the percolating septic tank effluent, the affinity of these sites for NH_4-N, the cation composition of the effluent, and the degree of saturation of the exchange sites with ammonium.

Nitrification, the biological transformation of NH_4-N to NO_3-N, is the most important nitrogen reaction occurring in soils, as NO_3-N does not react with the soil, but rather passes readily through a soil with percolating water. Nitrification is an aerobic reaction performed by obligate autotrophic organisms (Alexander, 1977). All of the nitrogen entering a soil absorption field (minus the 1-3% that enter a stable organic state) is presumed eventually to undergo nitrification to NO_3-N (Magdoff et al., 1974). The only significant loss of NO_3^- after this point will be due to denitrification.

Denitrification is the reduction of NO_3-N to nitrogen gas (N_2) and nitrous oxide (N_2O), is an anaerobic bacterial process wherein the presence of an energy source (organic carbon compounds) NO_3-N serves as the terminal electron acceptor in the absence of oxygen i.e. under saturated conditions (Sikora and Keeney, 1975). Thus, before denitrification can occur, nitrification must take place to provide the NO_3-N substrait. Thus, of the total nitrogen present in septic tank effluent, 1-3% goes into stable organic matter, a certain percentage is denitrified, and the rest can travel through the soil to the groundwater as NO_3-N.

Other nitrogen reactions, such as NH_3 volatilization, NO_3^- adsorption, and chemodenitrification as well as nitrogen uptake by plants in NO_3^- or NH_4^+ are not significant as their occurrence is minimal compared to the total nitrogen input by septic tank effluent (Sikora and Corey, 1976).

Phosphorus

The primary form of phosphorus in septic tank effluent is orthophosphate (85%), the rest being organic and condensed phosphates (Magdoff et al., 1974; Otis et al., 1975). Organic and condensed phosphorus will eventually be converted to ortho-phosphate over time (Sikora and Corey, 1976). Ortho-phosphate can participate in three types of transformation under a soil absorption field: chemisorption reactions, precipitation reactions (Sikora and Corey, 1976), and immobilization by microbial populations.

In chemisorption reactions, phosphorus forms stable compounds with iron, aluminum, and calcium minerals in the soil. Iron and aluminum phosphorus compounds predominate in acid to neutral soils, calcium phosphates in neutral to alkaline soils. Factors governing chemisorption of phosphorus include temperature, pH, cations in the soil solution, and the presence of organic anions capable of competing for bonding sites with phosphorus (Sikora and Corey, 1976). The presence of organic anions formed by degradation of organic matter could be significant below soil absorption fields receiving septic tank effluent.

Precipitation of various phosphate minerals can occur under soil absorption fields. In soil, the high concentration of phosphorus in septic tank effluent is likely to form precipitates of dicalcium phosphate or octacalcium phosphate which will transform to hydroxyapatite over time (Lindsay and Moreno, 1960). Figure 4 (from Sikora and Corey, 1976) depicts the various phosphorus precipitates formed in calcareous soils. The predominant precipitate found in columns representing a soil absorption field is octacalcium phosphate (Magdoff et al., 1974).

Immobilization by microbial population of phosphorus is small, but evidence exists (Hannapel et al., 1963) that phosphorus movement in soil occurs primarily in organic forms and is dictated by the activity and movement of microorganisms in the soil. It is reasonable to conclude that because soils sorb inorganic forms of phosphorus so strongly, movement of phosphorus would correlate with the movement of bacteria which have immobilized the phosphorus.

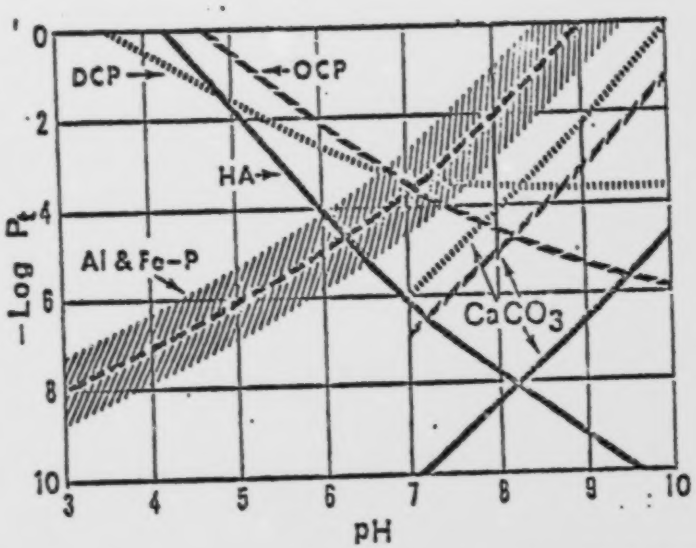

Fig. 7 Relationships between pH and soluble orthophosphate (P_t) for various inorganic phosphates including variscite (Al-P), strengite (Fe-P), dicalcium phosphate (CDP), octacalcium phosphate (OCP), and hydroxyapatite (HA). A Ca concentration of 10^{-3}M is assumed except for the curves in which solid phase $CaCO_3$ is controlling the Ca activity. For the $CaCO_3$ systems a CO_2 partial pressure of 3×10^{-4} atm is assumed.

from Sikora and Corey (1976).

Upon autolysis of these bacteria, organic phosphorus compounds will be released whose movement through soil is dictated by a different equilibrium than that governing inorganic phosphorus movement.

In addition, the microbial population of the soil and their byproducts (bacterial slimes) could serve to coat or shield soil particles that normally participate in chemadsorption and precipitation reactions. During testing of a system for removal of nitrogen and phosphorus from septic tank effluent, phosphorus removal was impeded by slime growth and organic anion competition for the sorbtion sites on the limestone used (Sikora et al., 1976).

Under anaerobic conditions, the soil solution phosphorus increases as the solubility of the iron compounds, which dictate the phosphorus equilibrium, convert from insoluble Iron (III) to soluble Iron (II). The phosphorus released into solution establishes a new equilibrium (Sikora and Corey, 1976) with Al and/or Ca bound phosphates. Extractable P in soils increases rapidly as redox potential decreases from +200 to -200 mv with the change from aerobic to anaerobic conditions (Patrick, 1964).

GROUNDWATER QUALITY

Nitrogen

Nitrate leaching through the soil and entering groundwater has two possible deleterious effects: eutrophication of surface waters, and health problems (methemoglobinemia) in infants. Nitrogen and phosphorus are considered to be the likeliest limiting nutrients in the primary productivity (algae and weed growth) of surface water (Steward and Culp, 1967). Nitrate entering surface waters from groundwater could be a factor in eutrophication of surface waters. Studies done in situ on waste disposal systems have indicated that a major source of NO_3-N in groundwater is septic tank effluent (Pruel, 1966; Polkowski and Boyle, 1970; USGS, 1979).

The World Health Organization and the United States Public Health Services have established a limit of 10 mg NO_3-N/L on potable water supplies. Bosh et al. (1950) attributed some cases of methemoglobinemia in infants to NO_3-N originating from septic tank effluent. Of 139 cases of methemoglobinemia, 129 were served by private wells with concentrations on NO_3-N greater than 10 mg NO_3-N/L; a majority of these wells contained coliforms (45 of 51 samples) indicating possible contamination by fecal wastes, possibly septic tank effluent.

As indicated above, the total nitrogen input in a soil absorption field minus that removed by fixation and denitrification may be nitrified and enter the groundwater. After it enters the groundwater, the only decrease in NO_3-N will be due to dilution by groundwater of lower concentration (Walker et al., 1973).

Phosphorus

Phosphorus movement through soil is thought to be minimal due to soil's ability to strongly sorb phosphorus (Jones and Lee, 1977). Phosphorus is sorbed so strongly that in a study of the water nutrient budget for Crystal Lake, Michigan, the Environmental Protection Agency concluded that phosphorus leaching from soil absorption fields comprised only 6% of the total phosphorus input into the lake (EPA Krause et al., 1979). In a study of phosphorus movement from septic tank effluent into groundwater, Jones and Lee (1977) concluded that no phosphorus was leaving the soil absorption area; thus, this author feels that phosphorus leaching from a septic tank system does not contribute to eutrophication of surface waters, and that soil absorption systems perform excellent removal of phosphorus from wastewaters.

The prevention of bacterial and viral contamination of groundwaters is a prime function of a subsurface soil absorption system. Many epidemiological studies have attributed disease outbreaks to contamination of water supplies by subsurface waste disposal systems. Bacterial and viral travel from a private waste disposal system is primarily governed by the type of water flow and the soil type and temperature. The mound systems must be able to prevent bacterial travel into groundwater aquifers by maintaining unsaturated flow from the systems; this aspect is particularly important for Package 2 (over creviced bedrock) and Package 3 (high water tables) designs.

MATERIALS AND METHODS

Study Design

In October 1976 approximately 260 mound systems were in operation in Wisconsin. Of these, 26 were randomly selected for monitoring. Nineteen systems were on sites with seasonally high groundwater (Package 3); 5 on sites with shallow bedrock (Package 2); and 2 on sites with slowly-permeable soils with and without high groundwater (Package 1). After one year, 3 systems, one from each Package type, were eliminated from further consideration because they were seldom used, and 10 more systems were added (total of 33 systems). The 10 additional systems included 2 Package 1, 2 Package 2, and 6 Package 3 systems (1 with a siphon instead of a pump). The proportions of the systems selected reflect the ratios of the 3 types of systems in use at each time the selections were made. Each system was given a code number indicating the system type (Package 1, 2, or 3) and when it was constructed on a progressive time scale. Figure 5 shows the locations of the mounds monitored. As of December 1978, 640 mounds had been installed in the state (Miller, 1978).

In November 1976, preliminary samples were collected to investigate various sampling procedures and analytical methodologies. Starting in February 1977, a regular bimonthly sampling and monitoring program was begun on each mound system chosen for study and continued until August 1978.

Septic tank effluent samples were taken from the pumping chamber,

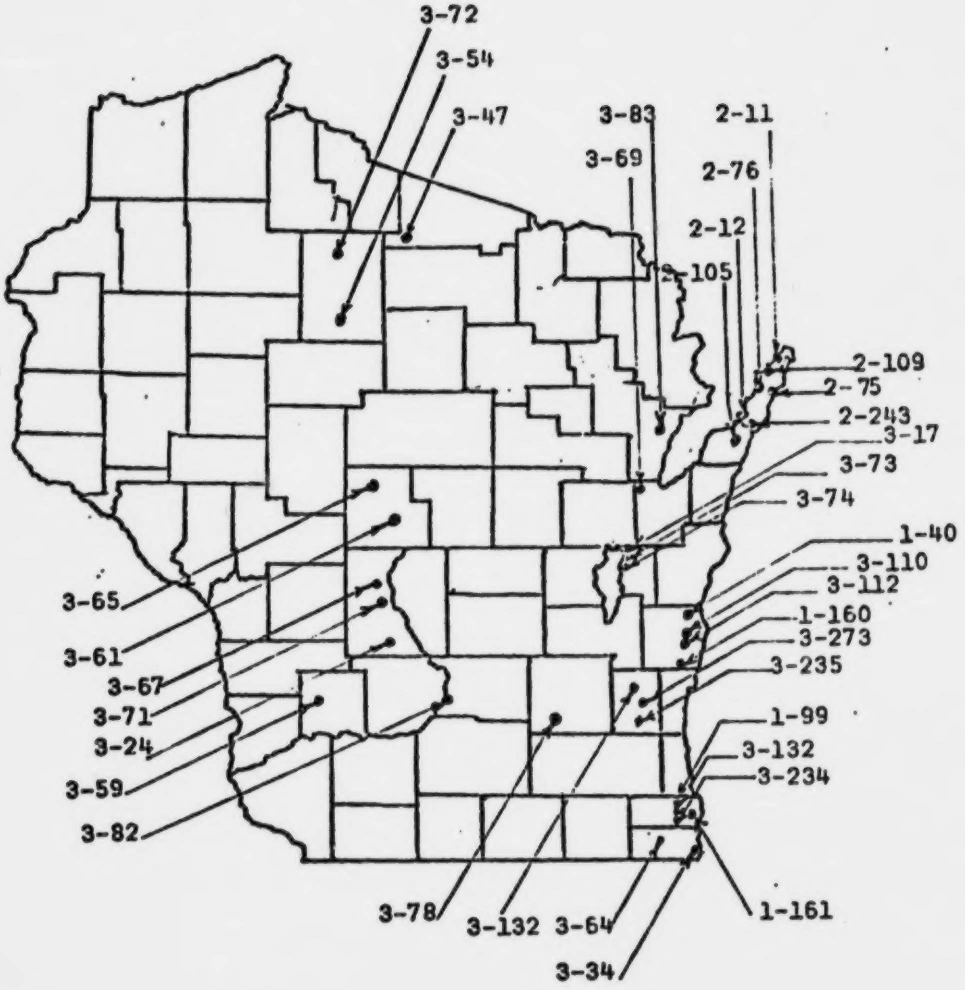
3-72
3-54
3-47
3-83
2-11
3-69
2-76
2-12
2-105
2-109
2-75
2-243
3-17
3-73
3-74
1-40
3-110
3-112
1-160
3-273
3-235
1-99
3-132
3-234
1-161
3-65
3-61
3-67
3-71
3-24
3-59
3-82
3-78
3-132
3-64
3-34

Fig. 5. State-wide locations of the mound systems studied.

soil samples were taken at various locations within and directly below the mound and from the surrounding area for comparison. Wherever possible, groundwater samples were obtained from wells installed downslope from the toe of each mound. In April 1978, new groundwater wells were installed around each of 4 mounds with high groundwater. These wells were sampled every 2 weeks until September 1978. All samples were brought to the laboratory in a refrigerated condition within 24 hours. The following parameters were chosen for study.

1. General Features
    a. Appropriateness of design, placement, and size.
    b. Wastewater loading rates.
    c. Fill material texture and particle size analysis.
    d. Retention of water within the mound; ponding in the absorption bed or at the natural soil surface.
2. Wastewater Characteristics
    a. Bacteriological [total coliforms (TC), fecal coliforms (FC), fecal streptococci (FS), *Pseudomonas aeruginosa (Ps. a)*].
    b. Chemical
        1. Nutrients [total Kjeldahl nitrogen (TKN), ammonium nitrogen (NH_4-N), nitrate nitrogen (NO_3-N), total phosphorus (TP), dissolved orthophosphorus (OP)].
        2. Biological oxygen demand (BOD), chemical oxygen demand (COD) electrical conductivity (EC), chloride (Cl^-), cations, pH.

   c. Physical [temperature, total solids (TS), volatile solids (VS), total suspended solids (TSS), volatile suspended solids (VSS)].

3. Attenuation Levels within the Mound and in Groundwater
   a. Bacteria [TC, FC, FS, *Ps. a*].
   b. Chemical parameters
      1. Nutrients [TKN, NH_4-N, NO_3-N, TP, OP].
      2. pH, total organic carbon (TOC), Cl^-, EC.

These tests were intended to evaluate the construction and performance of the mounds and the prospects for their continued satisfactory performance.

Sampling Techniques

General features

At each site, the native soil was examined by augering and checked against the soil descriptions given in SCS soil survey maps and certified soil tester reports. The dimensions and elevations of each mound and absorption area were measured using surveying equipment. A steel prod was used to locate the edges of the gravel in each absorption area.

To assess the wastewater load, each pumping chamber was calibrated to obtain a dose volume. The dose volume was determined by triggering a dosing event, and then refilling the chamber with a measured volume of water until another dosing cycle occurred. An electrical counter was installed in the pump control circuit to record the number of doses with time. The counters were sealed within a plexiglass box and positioned for easy inspection. However, condensation within the plexiglass box either corroded some counters or froze them during the winter months. Because of counter malfunctions, reliable flow or water use data were obtained from only 20 systems.

The Wisconsin Division of Health stipulates that each mound have observation pipes located at the bottom of the absorption bed on the gravel/fill material interface to reveal whether septic tank effluent ponds in the absorption bed. Such ponding may indicate uneven distribution of the effluent and/or a slow infiltration rate into the sand fill (Bouma et al., 1974a). During each visit to each mound system, the whole area was examined to determine if any surface discharge of untreated effluent had occurred; the presence and depths of any water standing in the observation pipes was noted and measured.

County code administrators, sanitarians, and plumbing inspectors were requested by newsletter and direct communication to supply information on any mound systems installed in their jurisdiction that experienced problems. Also, copies of all county inspection reports were obtained from the Wisconsin Division of Health. None of these gave any indications of system failures or malfunctions other than those noted below (see p. 60).

Effluent sampling

Effluent samples were initially obtained from the pump chamber with a 2-liter stainless steel bucket attached by a hinged bracket to the end of a collapsible aluminum pole. This allowed samples to be obtained which were representative of the effluent to be introduced into the mound and avoided contamination with any sludge collecting in the bottom of the pump chamber. The bucket was sterilized before each sampling by soaking it with ethanol and igniting the ethanol. Later a pole with a bracket to hold a 12 oz. sterile WHIRL-PAK® (obtained from NASCO Inc.) bag was used to obtain effluent samples.

Effluent samples were divided into two groups, and refrigerated for transport. Samples for bacterial, viral, BOD, and COD testing were placed in sterile WHIRL-PAK bags. Samples for chemical analysis were placed in 500 ml Nalgene bottles and treated with 1 ml/L of 0.1% phenylmercuric acetate in dioxane solution to prevent chemical transformations due to microbial action. All samples were transported in a refrigerated condition to the laboratory within 24 hours for analysis.

Soil sampling

To obtain soil samples from within and below the mounds, a special cased auger was used which allowed for accurate location of the sampling depths within the mound and prevented contamination of the samples with overlying material. This auger also allowed for exact reconstruction of the mound to prevent internal slumping or collapse and any short-circuiting of the treatment depth (Fig. 6).

The auger casing was made of 5 feet of 4-inch diameter thin-walled (1/32 inch) aluminum pipe fitted with a flange at the top to hold the auger handles while the hole was being drilled, allowing both the auger and the casing to turn simultaneously. The flange was also used to attach a length of pipe in place of the auger handle to withdraw the casing slowly after use to allow exact reconstruction of the mound as the casing was being withdrawn.

Two types of augers were used in conjunction with the casing. A

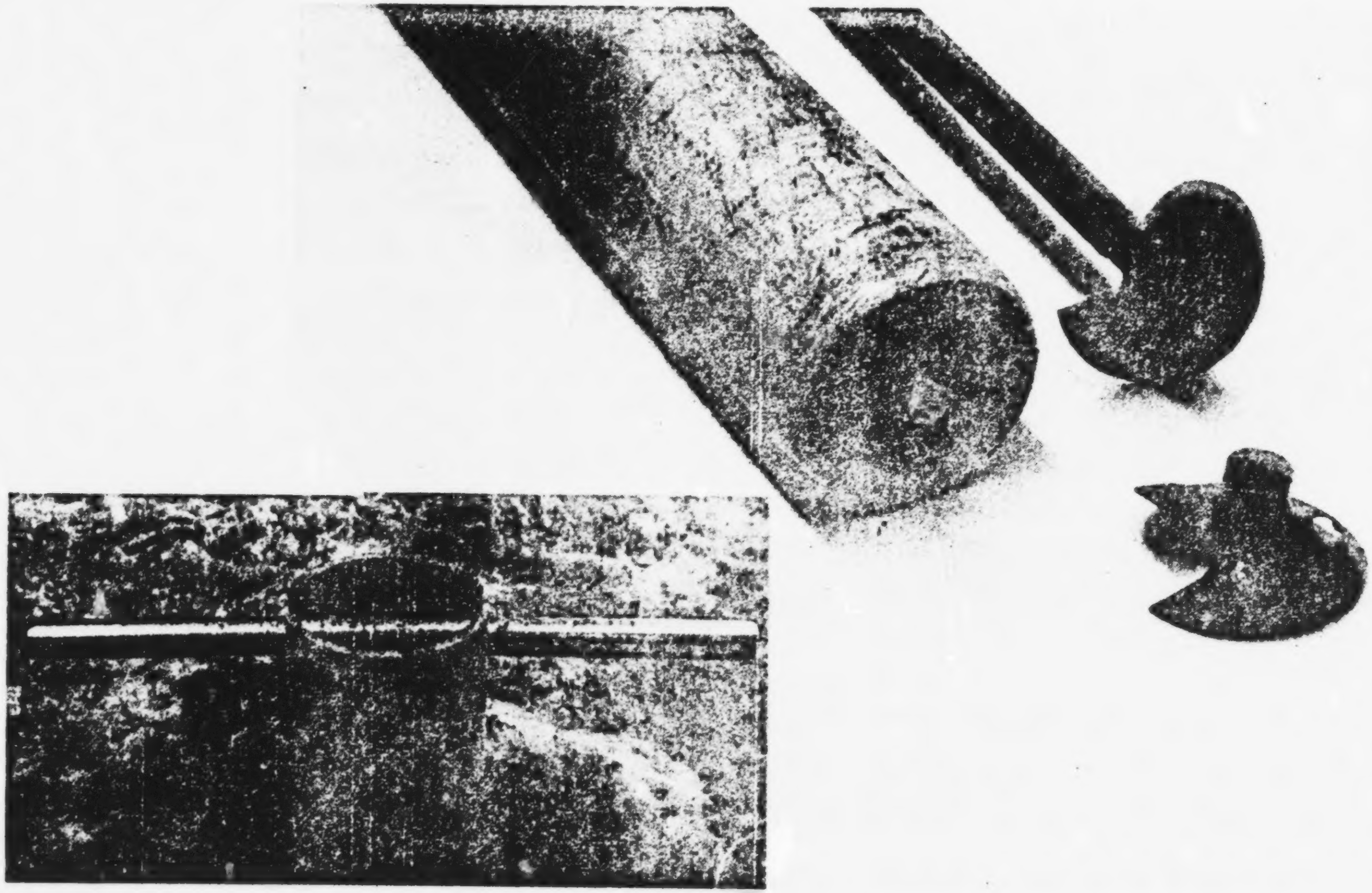

Fig. 6. Casing and auger used to obtain soil samples within and below the mounds.

conventinal 3-inch diameter bucket auger with a handle 4 feet long was used for digging through the cap or cover, fill material, and the soil below the mounds. To dig through the absorption bed with the aggregate, a special screw auger was constructed. The screw auger was sized to fit inside the 4-inch diameter metal casing with 1/16 inch of clearance, and to extend 1/4 turn below the end of the metal casing. The screw had only 1 flight with the pitch varying according to the size of the aggregate in the mound. The pitch for picking up 3/4 inch aggregate was approximately 1 inch, for pea gravel approximately 1/2 inch. The screw auger in conjunction with the casing allowed a cased holed to be dug perpendicularly through the gravel in the mound adsorption bed and samples to be removed at specific depths in the treatment area of the mound.

Soil and fill samples obtained for bacteriological analysis were taken using a special cased push auger (Fig. 7) that collected samples in the bottom of the casing without the possibility of contaminating the sample with material from overlying layers. The push auger was cased with a copper sleeve and sterilized prior to sampling by dousing and flaming with ethanol. All soil from fill samples were placed in WHIRL-PAK bags, immediately refrigerated and brought to the laboratory within 24 hours for analysis. Whenever possible, samples were collected at depths of 0-4, 4-8, 12-16, and 20-24 inches into the natural soil; for convenience in reporting, these depths are described as 5, 15, 35 and 55 cm, the midpoint of the depths sampled, in figures and tables.

During the digging of the sample hole, all materials taken from within and below the mound were placed separately on a plastic sheet.

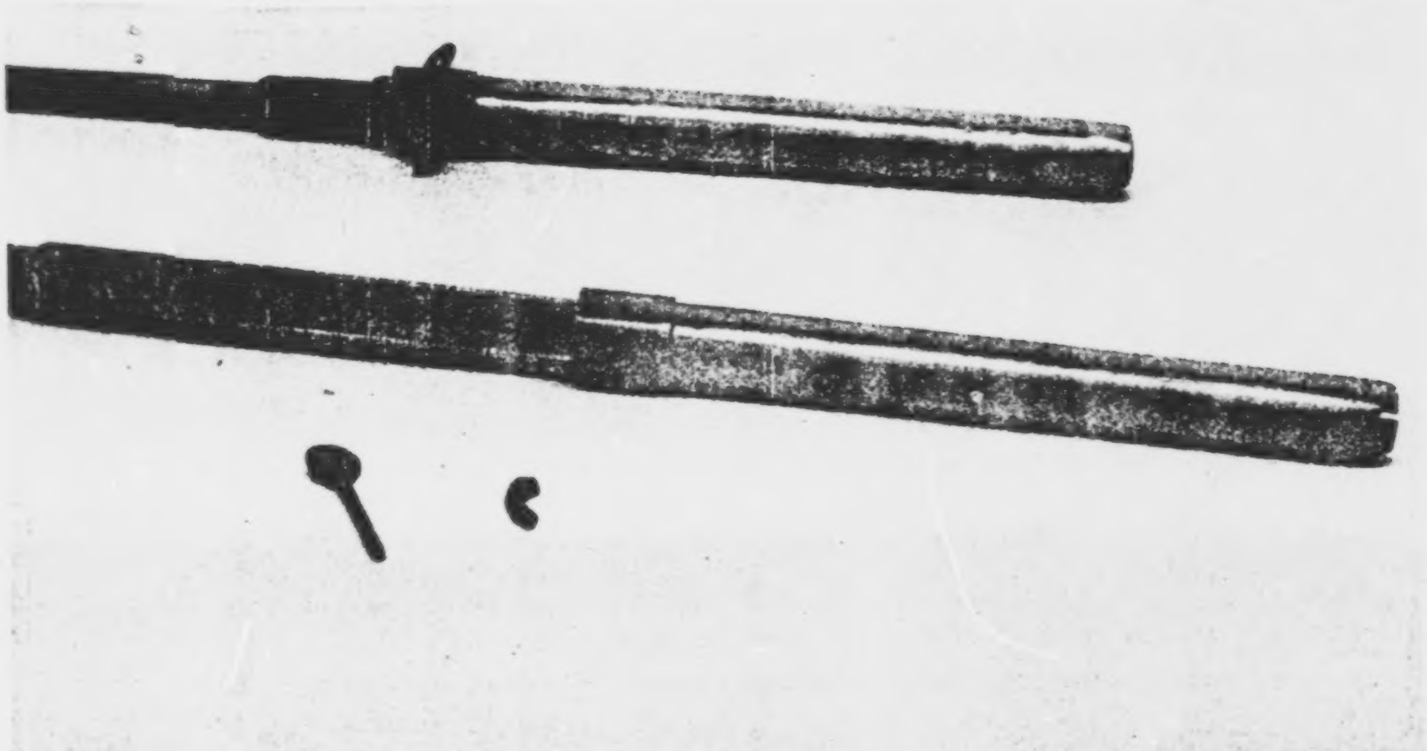

Fig. 7. Cased push auger used to obtain soil samples for bacteriological analysis within and below the mounds.

After sampling, as the casing was removed gradually from the hole, the material taken from the mound was returned to the hole in reverse order from which it was removed. This reconstituted the layers of soil in the mound just as they had been before sampling and prevented any short-circuiting of the mound and any damage to the physical appearance of the mounds.

During sampling, the auger sometimes encountered a transiently saturated zone (i.e. the auger hole filled with water) because sampling was being done soon after a dosing event or because groundwater was encountered at depth under Package 3 systems; digging was therefore stopped at that point to prevent the flow of untreated septic tank effluent through the auger hole. When only the absorption bed was ponded (i.e. the observation holes had standing water), soil samples were taken at an angle from the side of the mound; however, when the fill material or natural soil was saturated, no samples could be taken from below the level of saturation. This has led to some gaps in the monitoring data.

Groundwater sampling

Two types of groundwater sampling wells were installed during the mound monitoring program; both were constructed using schedule 40, 1-1/2 inch diameter PVC pipe. The wells used from February 1977 until April 1978 were open-ended PVC pipe perforated with 3/16 inch holes. They were installed by augering down to the aquifer, placing the pipe in the hole and refilling the hole with soil. The use of these wells was discontinued for two reasons. First, the wells did not have a satisfactory seal, so that seepage of surface waters down the side of the pipe contaminated the groundwater. Second, samples obtained from these wells contained large amounts of solids and did not give a true picture of the groundwater quality, but rather that of a soil-groundwater combination.

In April 1978, four mound systems with high groundwater were chosen and wells of modified design were installed near them. The new wells had four 1-foot rows of 3/16 inch holes drilled 1 inch apart covered by a layer of Dupont Style 3404 Typar® (4 oz/yd^2) which was glued into place. Typar is a spunbonded polypropylene filter fabric with an effective pore size of 120 μm. The wells were driven into the soil to a minimum of one foot below the water table to insure that samples could be obtained from the wells throughout the year. The samples obtained from wells coated with Typar contained practically no solids and therefore much less of the chemical species that are sorbed onto soil particles.

Holes augered for the old groundwater wells went only to the top of the groundwater because soil collapsed into the hole while augering into the waterlogged soil. Soil also washed into the wells while they were in position. Because the groundwater depth fell during the year, the old wells were often dry and no samples could be collected. The new wells were therefore fitted with a hardwood well point at the end of the PVC pipe (Fig. 8). The well point overlapped the outer edge of the PVC pipe by approximately 1/8 inch; this lip protected the Typar filter from tears as the well was driven into the aquifer. With the drive point, the well could be driven well below the surface of the groundwater.

For the new wells an adequate seal was provided around the well to

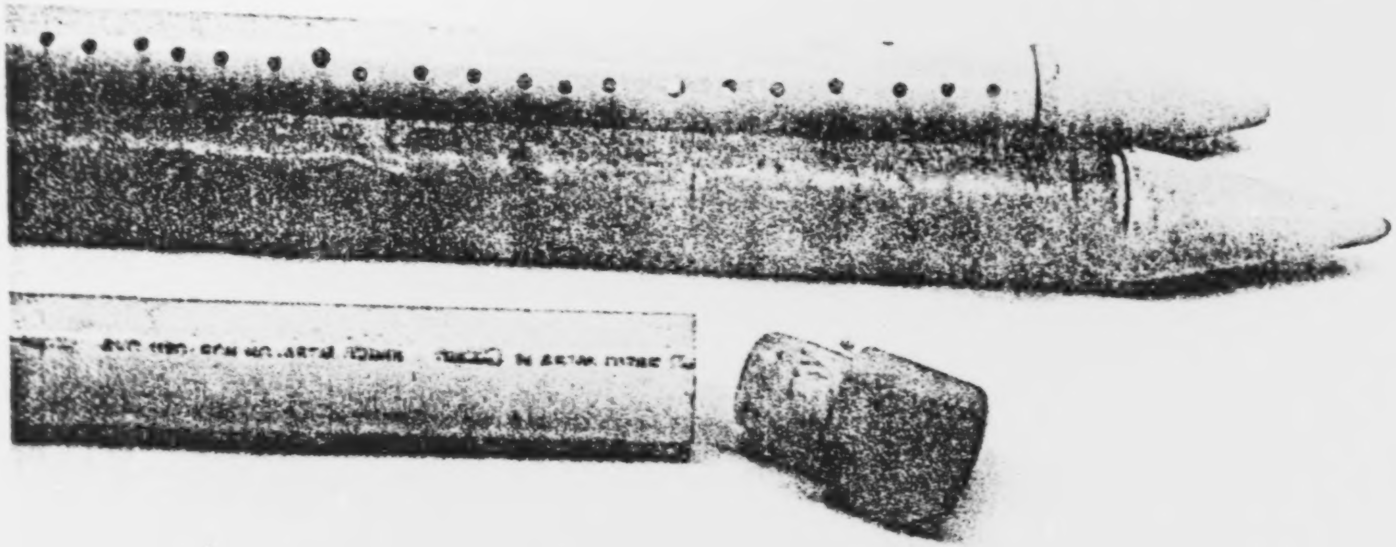

Fig. 3. Novel well design used for sampling groundwater near mounds.

prevent contamination of the groundwater by seepage of surface water. The hole around the pipe was backfilled to a depth of 1 foot from the soil surface with the soil removed from the original hole, tamping down the soil as each increment was added. A slurry of Durobond® joint sealing compound (United States Gypsum; Chicago, Illinois) was then poured around the well to about 2 inches from the soil surface; the rest of the hole was filled with sod. The slurry hardened round the pipe, embedding it tightly in the soil with a leakproof seal. All wells were capped with a threaded PVC fitting and closed with a PVC cap.

Samples were obtained from the wells using 3/16-inch I.D. Tygon tubing connected to a 1 L vacuum flask which was evacuated using a brass bilge pump. The flask and suction tube were rinsed twice with sterile distilled water and five times with water from the well to be sampled prior to each sampling. Before the sample was taken each well was either pumped to dryness and allowed to recharge or pumped to remove at least four times the volume of water in the well, to insure that the sample obtained was groundwater and not stagnant water that had been standing in the well. Samples for bacteriological analysis were immediately refrigerated and transported to the laboratory within 24 hours for analysis. Samples for chemical analysis were preserved with 1 ml/L of a 0.1% PMA solution, refrigerated, and transported for analysis.

Chemical methods

Total Kjeldahl nitrogen (TKN) was determined using the semimicro Kjeldahl procedure described by Bremner and Keeney (1966). This method involves the digestion of 0.3 g of air-dried soil, 5 ml of effluent or 20 ml of groundwater with 3 ml of conc. H_2SO_4 with 1.1 g of a potassium sulfate-catalyst mixture. Determinations were made by adding 15 ml of 10*N* NaOH and steam distilling the NH_3 formed into a boric acid indicator solution, and titrating with standardized H_2SO_4.

Ammonium and nitrate nitrogen (NH_4-N, NO_3-N) were determined using direct steam distillation of an extract of 10 g of wet soil, 5 ml of effluent or 20 ml of groundwater as described by Bremner and Keeney (1965). The wet soil was first extracted with 50 ml of 2*N* KCl, a 5 ml aliquot of the extract was treated with 1.1 g of MgO to release NH_3 which was distilled into 5 ml of a boric acid indicator solution. The addition of Devarda's alloy reduces the NO_3-N to NH_3 which was then collected in the boric acid indicator. Both solutions were then titrated with standardized H_2SO_4.

Total phosphorus (TP) was determined by digestion of 0.5 g of air-dried soil, 1 ml of effluent, or 1-5 ml of groundwater using the perchloric acid procedure; detection was performed using the ascorbic acid technique (Standard Methods, 1975).

Dissolved orthophosphate (OP) was determined by extraction of 2.5 g of wet soil, 2.5 ml of effluent or 2.5 ml of groundwater with 25 ml of a 0.001*M* $SrCl_2$ solution containing 15 g/L of Superfloc 214®, an acrylamide-acrylic flocculating resin produced by American Cyanimide Inc., New Jersey. The extract was filtered through acid washed Whatman #40 filter paper. Determinations were made by the ascorbic acid procedure described in Standard Methods (1975) using a 680 nm wavelength (on account of the Superfloc 214). This procedure is described by Wendt (1974).

Total organic carbon (TOC) was measured using a Leco Carbon

Analyzer, model #572-100: 0.2 g of air-dried soil was ignited in the induction furnace in a pure oxygen atmosphere and the resultant CO_2 was absorbed in KOH solution. The difference in volume in the KOH gives a direct readout in % C (Leco Carbon Analyzer, 1959; Tabatabai and Bremner, 1970; Young and Lindbeck, 1964).

Chloride (Cl^-) was determined using a Buchler-Cotlove Chloridometer Titrator, model #4-2000, on unfiltered wet soil and liquid samples. This instrument operates on the basis of a coulometric-amperometric titration with silver ions which are precipitated in an insoluble form and consequently inactivated by any chloride present in solution (Cotlove et al., 1958).

Electrical conductivity (EC) was found using a Yellow Springs Instruments model 31 conductivity bridge on unfiltered water and soil samples. Soil samples were air-dried and diluted 5:1 with deionized water (Bower and Wilcox, 1965).

pH was determined using a combination glass-calomel electrode connected to a Corning model 130 pH meter. Soil samples were air-dried and diluted 2:1 with 0.01*M* $CaCl_2$ solution.

Soluble cations (Ca^{++}, Mg^{++}, K^+, and Na^+) in the effluent were measured using a Perkin-Elmer 306 Atomic Absorption spectrophotometer. Methods described in the instrument manufacturer's manual were used in all cases.

Total solids (TS), volatile solids (VS), total suspended solids (TSS), and volatile suspended solids (VSS) were determined in the effluent samples according to Standard Methods (1975).

Biochemical oxygen demand (BOD) and chemical oxygen demand (COD) were determined on effluent samples according to Standard Methods (1975).

Physical properties

Particle size analysis (PSA) was performed on the fill material using the dry sieve technique and by hydrometer (Bouyoucos, 1962).

Fill material was sampled at five locations within each of the mound systems studied. The 5 samples were analyzed for particle size to obtain a uniformity coefficient and fines content for each fill material. The uniformity coefficient is a number used by civil engineers as a guide to the spread in grain sizes of soil materials; it is defined as:

$$\text{Uniformity coefficient} = \frac{\text{maximum diameter of finest 60\% of sample}}{\text{maximum diameter of finest 10\% of sample}}$$

The lower the uniformity coefficient, the more uniform the particle size and the more uniform the pore volume (spaces) between the soil particles. The mound construction recommendation for the texture of the sand fill was developed using a Plainfield sand (C horizon) with a uniformity coefficient of 1.8 and an average particle size > 0.25 mm (Bouma et al., 1973). A soil is defined as "uniform" only if the uniformity coefficient is less than 2.0 (Standard Specifications for Highway Materials,1955). A fill with a high uniformity coefficient and a small effective size (< 0.25 mm) would indicate that the fill is too fine.

Bacteriological determinations

Bacterial analysis on septic tank effluent and groundwater samples were performed using the multiple tube dilution or membrane filter techniques (Standard Methods, 1975) and expressed as most probable numbers per 100 ml of liquid (MPN/100 ml).

All soil samples were analyzed according to Standard Methods (1975) using 25 g of wet soil. TC, FC, FS and *Ps. a.* counts determined for soil samples were expressed as MPN per 100 ml of soil solution to simplify comparison with the counts in septic tank effluent and groundwater samples. Counts from samples from the same points of origin were grouped and averaged in terms of their geometric means. Attenuation ratios were calculated by dividing the bacterial counts in septic tank effluent by the counts in the sand fill or underlying soil. The medium for the *Ps. a.* test was modified by McCoy and Ziebell (1975) as follows:

Asparagine	2.0 g
Proline	1.0 g
K_2SO_4	10.0 g
$MgSO_4$	0.5 g
Ethanol, 95%	25.3 g
Distilled Water	1.0 l

Viruses in the effluent samples were determined by recent modifications (Cliver, 1975; Gerba et al., 1977) of the Standard Method (Standard Methods, 1975). This modified procedure allows for screening of a substantial number of enteric viruses including poliovirus, echovirus, and Coxsackie virus.

RESULTS AND DISCUSSION

Design Criteria

Adequacy of design criteria

Package type

Each system selected was surveyed and measured to ensure that it had been constructed according to package recommendations (Converse, 1976a-c) and with codes of the Wisconsin Division of Health (C. H62.20, Wisconsin Administrative Code, 1976). Three systems employing Package 3 designs (for permeable soils with high water tables) have been installed where the soil types call for a Package 1 design (for slowly permeable soils with high groundwater), because the county boards involved had not accepted the Package 1 design, but allowed installation of Package 3 systems. One system, 3-234, was located in a site different from that indicated in the plan approval and was improperly protected from runoff from a nearby hill; the homeowner had diverted a nearby stream to create an artificial pond, causing 3 inches of standing water around the downslope end of the mound. This mound was improperly installed by the homeowner (a local contractor) at a site of his own choosing, rather than where the plan indicated. No other irregularities concerning the site requirements and basic construction were found.

Water use and retention

Only 20 of the 33 mound systems monitored provided reliable counter data that allowed for accurate assessment of the wastewater flow. On the average, each of these systems produced a wastewater flow of 155.1

gal (587 L) per day. For the 19 of these 20 mound systems which served homes, the average family size was 3.97 persons. The average per capita water use in these 19 homes judged on the wastewater flow was 43.8 gal/day/person (Table 6). This figure is similar to that of other studies: 42.6 (Siegrist et al., 1976), 45.6 (Bouma et al., 1972), 44.0 (Bennett et al., 1975), and 43.0 gal/day/person (Witt et al., 1975). For these 19 homes the average water use was approximately 174 gal (658 L) per day.

The design criteria require that the wastewater estimate used to size a mound system be 150 gal/day/bedroom (C. H62.20, Wisconsin Administrative Code, 1976). Thus, at present the mounds are being overdesigned and, on the average, are operating at a much lower capacity than intended. This water volume provides for a large safety factor, but it should be possible to reduce the size of mounds while still maintaining an adequate safety factor and thus lower the cost of mound construction.

A measure frequently used by engineers in the design and operation of wastewater treatment systems such as sand filters and rapid infiltration systems is the loading rate, usually measured in gallons of wastewater applied daily to 1 square foot of absorption area (gal/day/ft^2). This is an appropriate parameter to use to design or examine the performance of systems that are in regular daily use. The loading rate concept has also been incorporated into the Wisconsin mound system design (Converse, 1978; Converse et al., 1976a-c).

The average loading rate for the 20 mound systems which furnished reliable flow data was 0.39 gal/day/ft^2 (arithmetic mean) or 0.33

Table 6. Water use characteristics for 20 Wisconsin mound systems

System	Total water use (gal/day)	Per capita water use (gal/day/person)	Loading rate (gal/day/ft^2)	Dose volume (gal/dose)	Dosing rate (gal/ft^2/dose)
1-40	241.0	48.2	0.51	437	0.89
1-161	82.7	20.7	0.22	214	0.58
1-160	90.0	30.0	0.25	45	0.12
2-11	129.6	32.4	0.35	271	0.75
2-12	135.3	67.6	0.52	167	0.65
2-76	212.0	42.4	0.58	205	0.57
2-109	159.0	53.0	0.43	325	0.81
3-34	10.2	5.1	0.05	362	0.97
3-47	79.0	39.5	0.21	353	0.95
3-61	87.6	29.2	0.23	53	0.14
3-64	569.0	114.0	1.16	515	1.05
3-69	139.0	46.3	0.37	354	0.96
3-71	92.9	30.9	0.25	378	1.02
3-72	43.3	21.7	0.12	425	0.65
3-73	145.4	48.3	0.39	340	0.92
3-74	217.0	72.3	0.58	390	1.05
3-78	84.0	*	0.12	425	0.65
3-82	274.6	68.5	0.74	318	0.86
3-83	91.6	18.3	0.18	77	0.16
3-110	219.0	43.8	0.44	388	0.79
n	20	19	20	20	19
Arithmetic mean	155.1	43.8	0.385	302	0.74

*A commercial establishment catering to various groups of people.

gal/day/ft^2 (geometric mean). Either value is well below the saturated conductivity of a heavily crusted medium sand (1.2 gal/day/ft^2) measured by Bouma et al. (1974b), and is therefore well within the design guidelines. Loading rates of this magnitude would assure proper performance of any mounds in continuous daily use, especially if applied in several portions, as the design criteria stipulate. However many mounds are actually receiving only intermittent use, or because of low use, are being dosed only every few days, and the low average loading rates measured for them do not reflect these use patterns. Some mounds are thus being overloaded sporadically with large dose volumes (gallons per dose), applied in a single dose. This is not apparent from a low loading rate measured over a long period of time. Thus, the loading rate is not an appropriate parameter to use as a guide in the design of mound systems receiving large sporadic doses of effluent. The low loading rate of systems not in constant use suggests that such systems maintain unsaturated flow conditions in the mound, but this is not so. The large dose volume (gal/dose) applied occasionally when such systems are in use momentarily far exceeds the recommended loading rate, creating flow conditions through the fill which are not conducive to good purification of wastewater.

Whether a mound is treating the wastewater volume/day while maintaining unsaturated conditions within the fill material can be better evaluated by measuring the dosing rate, which is the actual wastewater volume applied per dosing event divided by the area of the bottom of the absorption bed, or gal/ft^2/dose. Table 7 indicates the dosing rate and the estimated flow through the sand fill assuming that

Table 7. Estimated flow rate through a medium sand (cm/day) for various dosing rates assuming an infiltration time of 1 hour for each dose volume

Dosing rate		Flow rate (cm/day) for 1 hr infiltration time
gal/ft /dose	cm/dose	
0.24	1	24
0.48	2	48
0.72	3	72
0.96	4	96
1.20	5	120
1.44	6	144

the liquid volume from each dosing event infiltrates into the fill in 1 hour. This estimated average infiltration rate is based on times measured for infiltration following actual dosing events.

Bouma et al. (1972) determined that for maximum purification in a medium sand the flow rate should not exceed 40 cm/day. The average flow for an average dosing rate of 0.74 gal/ft^2/dose found in the 20 mounds (73 cm/day flow rate) is above the optimum flow that will allow for maintenance of unsaturated condition within the fill. Dosing rates exceeding 0.50 gal/ft^2/dose are likely to result in saturated flow through the fill material, judging from the observations of Bouma et al. (1972) on sands. Unsaturated flow conditions are essential for good chemical and bacterial renovation of wastewaters and to prevent escape of untreated wastewater from the downslope end of the mound.

During the monitoring period, 17 of the 20 systems were being dosed (volume per dose) at a higher level than that recommended in the design. The average dose volume was 308 gals/dose, versus the 200, 150, and 115 gals/dose recommended by Converse (1978) for 5-, 4-, and 3-bedroom houses. Table 6 shows the dose volume (gal/dose), dosing rate (gal/ft^2/dose), and loading rate (gal/day/ft^2) of the 20 mounds that furnished reliable flow data.

The effects of a high dosing rate are discussed later in connection with the results presented for bacterial and nutrient attenuation; however the high dosing rates are defeating the purpose of the distribution network and are promoting nearly saturated flow through the fill material after each dosing event.

Only one system (3-64) showed a combination of a high loading rate

(1.16 gal/day/ft^2) and a high dosing rate (1.05 gal/ft^2/dose); this mound was operating near the saturated conductivity (1.2 gal/day/ft^2) for a crusted medium sand fill found by Bouma et al. (1974). This system also had a sand fill with a uniformity coefficient of 9.9 and 15.7% fines (10.7% silt and 5.0% clay), indicating a USDA texture classification of a loamy sand.

The family using mound 3-64 comprised 5 people with an average water usage of 570 gal/day or 114 gal/person/day. No explanation could be offered by the homeowner for their high water use other than frequent entertaining on weekends.

System 3-64 exhibited ponded effluent in the gravel of the absorption bed during 4 of 9 visits and saturated conditions within the fill during 4 of 9 visits. The 4 incidents of saturated fill occurred in February, April, June, and August, 1977; the ponding situation was found in August, October, and December, 1977 and February, 1978. This suggests that system 3-64 may have begun to develop a crust in the absorption bed, allowing it to maintain unsaturated conditions in the sand fill. At no time during the two-year monitoring program did system 3-64 allow surface discharge of untreated effluent, indicating that even when a mound system is overloaded and has improper fill material it may still accept all the wastewater applied without allowing surface seepage.

During the monitoring program, each mound system was visited and sampled at least 6 times with a total of 347 visits to the 33 mounds being studied. Of the 33 mounds, 17 did not exhibit ponding in the gravel at any time inspected. Water was seen in the observation wells of 16 systems, indicating ponding in the gravel, a total of 29 times.

The fill material was also examined for signs of saturation. Fifteen systems did not show saturated conditions in the fill at any time during sampling. Only 1 of these 15 systems showed a ponded condition in the gravel. In the other 18 mounds saturated conditions in the fill were encountered only 27 times. Saturated conditions in the sand fill could indicate that the input of wastewater through the mound has exceeded the hydraulic conductivity of the natural soil, a condition that could lead to seepage of effluent out of the downslope end of the mound. In total, 11 of the 33 systems studied did not exhibit either ponding or saturated conditions in the fill at any time during sampling.

No mound ever showed a permanent state of ponding in the gravel absorption bed or permanently saturated conditions in the fill material. Interviews with the homeowners during site visits indicated that many incidences of ponding and/or saturated condition in the fill followed periods of excessive water use (i.e. large parties, extra laundry, etc.). Possibly some instances of ponding were encountered when a dose had just been pumped into the mound. A ponded situation after a dosing event need not be due to the presence of a crust impeding infiltration, but rather may indicate insufficient time for infiltration of the wastewaters into the fill.

Causes of hydraulic failure

Two mounds (3-110 and 3-112), both constructed by the same contractor, failed by surface discharge of untreated effluent during the winter of 1978. The average uniformity coefficient for these two mounds is 50.0. The fill used in these mounds was not sand but rather a loamy sand and sandy loam (USDA classification from soil texture triangle; U.S. Dept. of Agriculture, 1975). Both mounds were operating at a dosing rate (0.79 and 0.40 gal/ft^2/dose) above the maximum recommended for a medium sand (0.3 gal/ft^2/dose); because of the finer texture of their fill, these dosing rates are well above the infiltration rates of their respective fill materials. Both mounds allowed effluent to seep out of the mound at approximately the level of the gravel/fill interface, indicating that ponding was occurring in the absorption bed and the wastewater was surfacing before it entered the fill material.

During the summer the cover or cap on both of these mounds and the soil surrounding them was saturated, indicating that wastewater flow was occurring through the cap and not through the fill. In addition, in the summer of 1978, when samples were obtained from these two systems, the sand near the sand/gravel interface was wet, while the underlying fill material near the sand fill/natural soil interface was unsaturated or in many cases almost dry, indicating that little, if any, of the wastewater flow was occurring through the fill material.

During February 1977, system 1-160 failed. This mound met all the design requirements for mound construction (i.e., fill texture; uniformity coefficient = 2.2; loading rate = 0.25 gal/day/ft^2; dosing rate = 0.12 gal/ft^2/dose) except one; the cap or cover was insufficient. The lack of cover, landscaping, and snow on this mound allowed frost to penetrate to such an extent that the fill material became frozen and thus impermeable. The winter of 1976-1977 was one of the coldest in Wisconsin history (30 days of continuous temperatures below 0°C prior to February 1977), and many failures of conventional systems were reported by septic tank pumpers throughout the state during the same period. The mound that failed by freezing had no snow cover and was installed on an exposed site; this, in combination with the cold winter, caused the freeze-up.

Freezing of septic systems is usually unlikely because of the continuous influx of hot water from the household and the heat created by biological fermentations in the wastewater. Only two instances of transient system freeze-up were observed in a study of seepage pits (dry wells) in the permafrost region of Alaska (Hickey and Damien, 1966). Studies of the internal temperature of two mound systems in Wisconsin (Bouma et al., 1973) revealed that a mound in Door County, Wisconsin, which had an abundant snow cover, showed a range of +2°C to +17°C in and below the absorption bed (November to May). However, for another mound system without snow cover in Bayfield County, Wisconsin, the January to August temperatures in the absorption bed ranged from -1°C to +17°C, with a -2°C temperature occurring just below the topsoil cover (Bouma et al., 1975). Thus, it appears that low temperatures in combination with inadequate soil cover or snow cover could create problems for mound systems.

Starting in the fall of 1977, the temperature of the effluent in the pump chamber was measured (Table 8). These temperatures ranged from 0°C to 23°C; the frozen pump chamber occurred in a system that was not being used during the winter. The temperature of the water ponded in the two systems that failed in the winter of 1977 to 1978 was 2 to 2.5°C.

Thus, low temperatures appear to be a problem in a few mound

Table 8. Temperature, in °C, of effluent in pump chamber (33 mounds)

n	Arithmetic mean	Standard deviation	Standard error	Highest value	Lowest value	95% confidence limits	
141	12.7	6.0	0.5	23	0*	15.2	11.5

*The 0°C temperature occurred in a mound system that was not in operation during the winter.

systems. Insulation of the covers of the pump chamber (the most exposed component of a mound system) could help prevent cooling of the septic tank effluent. Proper landscaping of the mound with the recommended cover (insulation) is also extremely important and should not be taken lightly by either the contractor and/or homeowner. Schutt et al. (1979) suggested guidelines for locating and landscaping a mound system that emphasize this point and give recommendations for soil type, vegetation, etc. that are important in mound construction.

Wastewater Characteristics

Evaluation of the performance of the mound system required that the wastewater or septic tank effluent be characterized. Table 9 indicates the average characteristics for effluent samples taken from all the systems studied; a table of values for each individual system is available at cost from the Wisconsin Water Resources Center to those seeking additional information. A discussion of the importance of the various parameters follows. In general, the parameters were chosen to evaluate nutrient (nitrogen and phosphorus) attenuation, bacterial (potential pathogens) attenuation, and prospects for continued satisfactory performance of the mound system (solids, EC, BOD, COD, cations).

Nutrients

Nitrogen (N) is of primary importance for two reasons: first, as a plant nutrient, N in runoff or seepage to surface waters can contribute to lake eutrophication, and second, N transformations within a septic system absorption field can cause leaching of nitrate onto underlying groundwaters serving private and public wells. Nitrates in potable

Table 9. Average wastewater characteristics for septic tank effluent taken from 33 mound systems

Parameters	n	Arithmetic mean	Standard deviation	Standard error	Geometric mean
pH	152	7.3	0.37	0.03	7.3
EC (m mhos/cm)	207	909.8	890.8	61.9	748.9
Cl^- (mg/L)	189	164.3	363.7	26.4	75.2
TKN (mg/L)	127	81.5	58.0	5.15	69.9
NH_4-N (mg/L)	215	53.5	29.8	2.0	45.3
NO_3-N (mg/L)	215	0.95	3.3	0.23	0.008
TP (mg/L)	215	21.83	10.4	0.71	19.83
OP (mg/L)	215	17.56	18.4	1.25	14.4
BOD_5 (mg/L)	145	131.8	83.9	6.97	103.2
COD (mg/L)	52	445.0	177.6	24.6	404.6
TS (mg/L)	165	895.1	548.2	42.7	777.6
VS (mg/L)	164	305.5	162.8	12.7	277.9
TSS (mg/L)	164	87.4	57.0	4.45	78.9
VSS (mg/L)	164	66.0	52.1	4.07	64.8
Cu (mg/L)	36	0.09	0.11	0.02	0.08
Ca (mg/L)	17	91.65	36.33	8.82	88.49
Mg (mg/L)	52	25.1	16.84	2.34	22.15
Fe (mg/L)	35	0.69	0.97	0.16	0.73
K (mg/L)	52	20.0	9.8	1.36	17.4
Na (mg/L)	52	272.8	294.3	40.8	252.1
Zn (mg/L)	35	0.33	0.46	0.08	0.28
TC (MPN/100 ml)	205	8.7×10^6	2.5×10^7	1.7×10^6	2.1×10^6
FC (MPN/100 ml)	205	2.8×10^6	5.0×10^6	3.5×10^5	6.1×10^5
FS (MPN/100/ml)	198	2.5×10^5	1.3×10^6	9.2×10^4	8.5×10^3
Ps. a. (MPN/100 ml)	177	4.3×10^4	1.9×10^5	1.4×10^4	2.4×10^3

water supplies are limited by law (Safe Drinking Water Act PL 93-523) to less than 10 mg/L (as N), as high nitrates may lead to methemoglobinemia in infants (Comly, 1945; Gruener and Shuval, 1970; Shuval and Gruener, 1972; Delfino, 1977).

Nitrogen forms in domestic wastewaters are primarily ammonium-N (NH_4-N) and organically-bound N. Previous studies suggest an average N content in wastewaters of 40 - 80 mg/L (IES, 1979). In the mound monitoring project, the average concentration of N in the wastewaters was found to be 82.4 mg N/L with the average family of 3.97 persons using 658 L per person per day contributing a total of 19.8 kg N/yr to their soil absorption field or an average of 5.0 kg N/person/yr. An average protein consumption of 100 g/person/day would give a load of approximately 16 g N/person/day or 5.84 kg N/person/yr (Deutsch, personal communication). The figure of 5.0 kg N/person/yr reflects the fact that some homes were not occupied on a full-time basis.

The major sources of N to a septic tank are feces and urine which contain urea, uric acid, ammonia, undigested proteinaceous foodstuffs, and bacterial cells (Witt et al., 1975). The transformation of N upon entering an anaerobic septic tank begins when microbial enzymes such as proteases, ureases, and deaminases break down organically-bound N. Proteases hydrolyse proteins to short-chain peptides and amino acids. The peptides are further degraded to amino acids and deaminase enzymes release ammonia (NH_3) which at a pH below 7.8 become primarily ammonium (NH_4^+). Ureases break down urea and uric acid to release CO_2 and NH_3 (West et al., 1966). The resulting N products in the septic tank are thus predominantly ammonium and organic N; Otis et al (1975) give proportions of 75% and 25%, respectively, for these forms.

Analysis of septic tank effluent samples indicate that the N in monitored systems is primarily NH_4-N (64%), and organically-bound N (35%). Only 1% of the N in septic tank effluent is NO_3-N. This indicates that the pump chamber is anaerobic, that there is no net NO_3-N accumulation in the septic tank effluent, and that nitrate found in the soil absorption field is the result of nitrification of the N in the effluent under aerobic conditions. The NO_3-N in the septic tank effluent is probably due to a recent introduction of nitrate to the septic tank from the household, e.g. from nitrates in meat or vegetables.

The average pH of the septic tank effluent is important for many reasons (biological activity, precipitation reactions, possible loss of ammonia gas from the effluent, etc.). Above a pH of 9.25 the equilibrium between $NH_4^+ \rightleftarrows NH_3$ will favor NH_3 and may allow ammonia gas to leave the effluent by volatilization. The average pH of the effluent is 7.3, (Table 9) with the three highest systems having average pH levels of 7.9, 8.0, and 8.1. The homeowners indicated that this may be due to their use of caustic drain cleaning products (e.g. Drano).

Raising the pH of a septic tank above 7.8 need not produce beneficial results in the treatment of wastewaters. The small loss of NH_3 gas would reduce the nitrate leaching from a soil absorption system, but the effects upon the biological activity and chemical hydrolysis of fats, etc. could outweigh this benefit by causing premature failure of the soil absorption field. Studies of wastewater quality by Shannon (1975) suggest that the use of nonphosphate detergents (i.e. carbonate-built detergents) may cause high pH in some septic tanks and detrimentally affect system performance.

Phosphorus (P) in the phosphate form is a plant nutrient of primary concern in wastewater treatment. Phosphates entering surface waters, whether by runoff or seepage through the soil, can contribute to eutrophication.

Phosphorus in septic tank effluent originates from detergents, food wastes and human excreta. The relative amounts from each source is highly variable but detergent P inputs account for about 50% of the total P in domestic wastewaters (Sawyer, 1965; Hetling and Carcich, 1973). The contribution from human excreta is estimated from dietary data to be 0.23 to 1.05 kg/person/yr with a mean of 0.6 kg/person/yr (Sherman, 1952). Other estimates place the total waste effluent P contribution at 0.8 (Otis et al., 1975) to 3.0 kg/person/yr (Bernhard, 1975). In this study, the average total P concentration for the effluent was 21.8 mg/L; thus, with an average total wastewater load of 658 L/family/day, the average mound system contributes 4.73 kg of P/yr to the soil absorption field. Dividing this figure by the average family size (3.97 persons) indicates that the average P load is 1.32 kg/person/yr, a figure close to the average value of 1.5 kg/person/yr derived from several studies (Hetling and Carcich, 1973).

Under anaerobic conditions in the septic tank, organic and condensed phosphates are converted to soluble orthophosphate: 85% of the TP in septic tank effluent is orthophosphate (Magdoff et al., 1974;

Otis et al., 1975). Approximately 80% of the total P in septic tank effluent in mound systems is in the orthophosphorus or dissolved inorganic phosphorus form (OP). The remainder (organic P and condensed phosphates) is likely to be converted over time to orthophosphate. Studies with soil columns have shown the total P concentration in septic tank effluent to range from 15.6 to 25.4 mg/L with a mean value of 20.6 (Magdoff et al., 1974). A field study of 6 septic tank systems found the total P concentration to range from 11.0 to 31.4 mg/L with a median value of 12 mg/L (Otis et al., 1975). Mound monitoring showed the mean total P concentration in 33 septic tank to be 21.8 mg/L with a range of 5.5 to 43.7 mg/L of total P.

The average concentrations for some other effluent parameters monitored--electrical conductivity (EC), chlorides (Cl^-), BOD, COD, and cations--are summarized in Table 9. Some parameters were measured primarily to evaluate the performance of the mound system regarding groundwater modeling and continued satisfactory performance of each system, and are discussed later (p. 154).

The results of TS, VS, TSS and VSS analyses for 165 samples of septic tank effluent are surprising in comparison to values found in other septic tanks (Siegrist et al., 1976; Jawson, 1976). There is approximately a threefold increase in TS over the TS content in typical septic tank effluent (Table 9). This could be due to two factors. As the effluent enters the collection chamber, it falls from the pipe to the surface of the collected effluent. This drop ranges from 3 ft when a chamber is full (just prior to a dosing event) to as much as 8 ft in some systems. This fall could reintroduce oxygen into the effluent and promote bacterial growth, producing high VS values. Also, the pump used in the dosing system may resuspend solids from the sludge accumulating in the bottom of the collection chamber.

The high solids contents could result in more rapid soil clogging and crust formation within the mound (Jawson, 1976; Laak and Healy, 1974). However, during the two-year study no mound had developed a clearly recognizable crust and no system had failed due to excessive crust formation. Thus, it appears that the dosing system is functioning to prevent crust formation as it was designed to do.

Bacteria

A primary concern of any wastewater disposal system is the prevention of transmission of potential pathogens into the environment. A private septic wastewater system must prevent surface discharge of human wastewaters and the soil absorption field must be able to eliminate bacteria and viruses from the water before it enters groundwaters or creviced bedrock.

The bacterial flora of septic tank effluent are primarily those of intestinal or enteric origin. In warm-blooded animals, the intestinal flora are fairly stable and are not affected by the bacteria present in food consumed daily. The human intestinal tract offers an excellent environment for anaerobic bacteria (both spore forming and non-spore forming) and many other groups (Ziebell et al., 1974). In addition, if the host is currently infected, there will be pathogenic organisms in the raw effluent (e.g. *Staphylococcus aureus, Pseudomonas aeruginosa, Salmonella*, etc.). As the fresh raw effluent enters the treatment system (septic tank/soil absorption field) conditions change (aerobic to anaerobic) and new forms of bacteria can multiply to compete with the intestinal flora. The flora of sewage is a very complex but balanced mixed culture that is the primary agent in the biodegradation of the raw effluent. Thus, the microbial population of sewage is essential to the proper treatment of the wastewater.

While the level of treatment in a septic tank is significant, septic tank effluent still contains extremely high numbers of coliform bacteria, as well as streptococci, and possibly staphylococci, and salmonellae; thus, septic tank effluent must be considered unsafe for human contact (McCoy and Ziebell, 1975; Sauer et al., 1976). Table 6 summarizes the results of bacterial counts of septic tank effluent collected from all the mound systems studied. The results of bacterial analysis of septic tank effluent collected from mound systems compare well with those of other studies (Ziebell et al., 1975; Sauer et al., 1977).

The bacteriological parameters studied (TC, FC, FS, *Ps. a.*) were chosen to show the incidence and level of human waste contamination within and beyond the mound systems. The FC and FS are commonly used as indicators of recent fecal pollution and the possible presence of pathogens; these organisms are a part of human intestinal flora shed daily with feces.

The *Ps. a.* is also a common indicator of human fecal contamination as well as being an opportunist pathogen which can cause severe ear infections and enteric disorders in infants; it was present in all septic tank effluent samples analyzed, indicating that it is present in sufficient numbers to be significant, i.e. shed by many apparently healthy persons in high enough numbers to offer statistically valid recovery data (cf. McCoy and Ziebell, 1975 and Table 6).

Viruses

One of the major concerns in mound performance was the removal of viruses from septic tank effluent by the soil absorption field. In an effort to evaluate this parameter, septic tank effluent samples were obtained, stabilized by chilling (4°C) and returned to the laboratory for analysis of viral content by modifications (Cliver, 1975; Gerba et al., 1977) of the standard procedure (Standard Methods, 1975).

From Feb. to Oct. 1977, 78 samples of septic tank effluent analyzed showed no evidence of virus. Thus, to reduce the laboratory workload and to enhance the opportunity of finding viruses, samples for viral analysis were taken only from systems which served families with young children. Even then virus was isolated from only one system, recurring in three samples collected over a four-month period; this was the only virus found during the two-year monitoring program. One child in the family using the virus-infected system was being immunized against poliomyelitis with oral Sabin vaccine during the four-month period when a positive virus presence was found. No virus was ever found in the soil below this system.

Mound Performance

Bacterial attenuation

Introduction

In order to demonstrate removal of bacteria from septic tank effluent as it seeps through the sand fill within the mound and the layers of natural soil below, some common denominator must be used to compare counts obtained from liquid samples (septic tank effluent) and solid samples (sand and soil samples). Counts based on a dry weight basis for solid samples give a highly anomalous picture of bacterial movement for all the indicator species examined, both for individual systems and groupings of systems, because of the low moisture content encountered in many sand samples. However, in general, counts based on sand or soil moisture contents clearly reveal a gradual reduction in TC and FC numbers as effluent moves through the mound. Since bateria in soil are associated with the moisture films on and between soil particles, there is good justification for expressing bacterial counts in this manner. Moreover, counts of TC and FC based on soil moisture contents correlate much better with the Cl^- levels in soils than counts based on a dry weight basis. Since Cl^- is an ion species which remains unchanged in the soil solution and is not sorbed onto particles and since, on the average, Cl^- levels correlated well with sand and soil moisture contents and Cl^- levels in the effluent, the soil moisture was obviously derived principally from percolating effluent. Bacterial counts based on soil moisture therefore indicate how many bacteria are moving unhampered like Cl^- through the sand fill and underlying soil. The reduction in bacterial counts in soil solution at various levels can be expressed as attenuation ratios (bacterial counts in septic tank effluent divided by bacterial counts in soil samples). These are factors indicating bacterial die-off which reveal accurately how well overlying layers of sand or soil have removed bacteria as effluent seeps through the mound. Bacteriologists working with wastewater disposal normally measure bacterial die-off or removal in terms of log factors.

Log factors of 1, 2, 3 and 4 would correspond to attenuation ratios of 10, 100, 1,000 and 10,000.

Counts of FS and *Ps. a.* based on soil moisture continue to suggest an anomalous movement of these species through the soil (enrichment rather than reduction with depth) in some cases. Explanations for this behavior are given below.

All mounds

In general the mounds perform excellently in removing both TC and FC from the percolating effluent before the natural soil surface. The bacterial attenuation ratios (bacterial counts in septic tank effluent:bacterial counts in soil) shown in Tables 10-14 and the bacterial counts shown in Figs. 9-13 indicate that, on the average, TC are 99.53% and FC 99.9% removed before the effluent reaches the natural soil surface. All of the mounds studied performed as expected, the worst mound system (3-34) even removing 92.8% TC before the natural soil surface.

Coliform travel into the natural soil is also minimal; only system 3-34 allowed coliform travel into the natural soil. System 3-34 is over a Bogen loamy sand, a coarse-textured soil that will allow bacterial travel under saturated conditions.

Tables 10-12 and Figs. 9-13 show that in general the mound fill did not reduce the numbers of FS or *Ps. a.* to any great extent. In several systems, the FS and *Ps. a.* numbers at the natural soil surface and 5 cm into the soil are higher than the corresponding counts in the sands above and often in the effluent. This suggests that high effluent dosing rates allow these bacteria to travel through the sand fill and to

Table 10 Bacterial attenuation ratios (geometric mean of effluent MPN/100 ml geometric mean of soil solution MPN/100 ml); fill and soil texture (s = sand, l = loam, si = silt, = clay, etc. as per USDA soil classification); average fill and soil moisture contents (% H_2O by weight); loading rate (LR, gal/day/ft²); dosing rate (DR, gal/ft²/dose); dose volume (DV, gal/dose); uniformity coefficient (UC), average particle size (APS, mm) and particle size analysis (PSA, % by weight of standard sieve fractions as in Table 2) of the sand fill; and soil description for 14 individual mounds systems which furnished substantially complete sets of data (ND = insufficient data available).

Depth	Texture	H_2O, %	TC	FC	FS	Ps. a.
System 1-161:	LR 0.22; DR 0.58; DV 214; UC 2.2; APS 0.15 soil - Kewaunee silt loam, 2-6% slope, eroded/Ashkum silty clay loam, 0-3% slope PSA 0.22% 1.11% 2.73% 25.01% 51.09% 11.13% 8.37%					
Fill	s	11.4	1,041.9	5,461.2	1,586.3	0.90
5 cm	sil	32.6	132.2	1,477.9	3.1	0.32
15 cm	sil	24.3	520.9	11,915.5	2.1	0.88
35 cm	sic	22.3	2,530.4	38,550.0	7.7	1.75
55 cm	sic	24.8	2,108.7	11,915.4	1.7	1.91
System 2-11:	LR 0.35; DR 0.75; DV 271; UC 2.2; APS 0.25 soil - Longrie loam, 0-2% slope PSA 22.73% 8.53% 10.18% 21.32% 24.53% 6.88% 5.58%					
Fill	s	6.9	527.5	266.1	0.01	1.56
5 cm	l	44.8	831.1	4,323.8	0.22	45.1
15 cm	l	34.9	13,239.5	12,217.0	0.13	126.2
35 cm	l	26.9	3,385.8	5,524.0	0.12	ND
55 cm	l	21.8	2,738.8	4,469.0	0.10	ND
System 3-34:	LR 0.05; DR 0.97; DV 361.5; UC 7.5; APS 0.35 soil - Boyer loamy sand, 1-6% slope PSA 22.73% 8.53% 10.18% 21.32% 24.53% 6.88% 5.58%					
Fill	s	7.7	65.1	117.5	7.6	0.19
5 cm	ls	4.6	13.8	143.1	0.001	0.05
15 cm	ls	10.9	169.7	334.8	2.1	0.26
35 cm	sl	5.7	174	174.5	0.10	0.15
55 cm	sl	6.1	17.6	186.7	0.05	ND
System 3-47:	LR 0.21, DR 0.95; DV 353; UC 3.7; APS 0.3 soil - Alcona fine sandy loam PSA 6.44% 4.34% 12.47% 32.07% 32.66% 7.75% 4.16%					
Fill	s	11.6	40.2	3,680.7	0.11	4.6
5 cm	sl	15.6	41.1	380.4	1.47	0.76
15 cm	sl	21.0	543.0	6,657.6	0.67	8.9
35 cm	sl	14.4	513.0	4,365.2	1,127.4	6.7
55 cm	sl	13.6	728.4	4,321.1	5.3	ND
System 3-61:	LR 0.23; DR 0.14; DV 53; UC 5.4; APS 0.5 soil - Eauplaine silt loam, 2-6% slope PSA 19% 12% 25% 34.5% 6% 0.59% 2.5%					
Fill	s	6.3	5,267.4	915.6	62.4	0.07
5 cm	sil	40.7	523.8	2,575.2	85.9	0.04
15 cm	sil	33.0	40,000.0	5,693.3	182.3	0.33
35 cm	sil	25.0	32,886.1	4,814.2	390.6	ND
55 cm	sil	19.0	24,993.8	3,658.7	296.8	ND
System 3-66:	LR 1.16; DR 1.05; DV 515; UC 9.9; APS 0.6 soil - Morley silt loam, 2-6% slope PSA 26.24% 7.74% 11.33% 27.23% 35.36% 4.24% 3.09%					
Fill	ls	23.6	1,132.4	798.3	181.2	1.99
5 cm	sil	35.8	192.5	2,794.2	11.7	0.34
15 cm	sil	30.7	3,457.7	6,359.3	13,280.6	2.1
35 cm	sic	27.7	15,655.1	9,009.9	ND	ND
55 cm	sic	19.3	ND	ND	ND	ND
System 3-69:	LR 0.37; DR 0.96; DV 354; UC 3.2; APS 0.3 Soil - Oshkosh sandy loam, 2-6% slope, Solona loam, 1-3% slope PSA 11.06% 7.74% 11.33% 27.23% 35.36% 4.24% 3.09%					
Fill	s	13.3	93.1	159.9	5.7	0.36
5 cm	sl	22.2	194.8	299.8	2.07	1.19
15 cm	sil	21.0	1,473.7	6,720.2	27.2	3.5
35 cm	sic	23.0	1,011.8	455.7	4.8	1.03
55 cm	sic	19.3	72,135.7	25,184.6	27.9	ND

Table 10 (cont.)

System 3-71: LR 0.25; DR 1.02; DV 377; UC 20; APS 0.18 System 3-71: LR 0.25;
DR 1.02; DV 377.5; UC 20; APS 0.18
soil - Billet sandy loam, 2-6% slope
PSA 1.16% 0.43% 3.64% 25.3% 61.44% 5.60% 2.70%

Fill	s	6.4	82.8	882.2	4.3	0.72
5 cm	sl	7.2	349.6	528.8	3.0	0.14
15 cm	sl	11.1	681.2	1,530.4	5.7	1.22
35 cm	sl	12.8	156.1	ND	11.3	ND
55 cm	sl	15.9	194.1	ND	14.0	ND

System 3-72: LR 0.12; DR 0.76; DV 281; UC 6.4, APS 0.6
soil - Pence loam, 2% slope
PSA 27.76% 10.12% 16.14% 27.38% 12.58% 1.27% 4.64%

Fill	s	7.1	2,212.4	170.8	0.62	1.50
5 cm	f	44.4	822.1	229.1	4.9	0.58
15 cm	f	33.8	13,490	2,224	770.2	15.0
35 cm	f	23.6	26,011.2	20,080	9.0	ND
55 cm	f	ND	ND	ND	ND	ND

System 3-73: LR 0.39; DR 0.92; DV 340; UC 3.1; APS 0.2
soil - Kewaunee silt loam, 2-6% slope
PSA 1.20% 1.18% 4.46% 32.04% 45.04% 9.50% 6.48%

Fill	s	12.0	95.4	23,236.5	14.1	2.03
5 cm	sil	28.1	188.4	357.1	22.9	0.59
15 cm	sil	30.6	4,625.7	9,583.1	24.7	3.9
35 cm	sl	29.5	1,073.1	71.5	34.8	ND
55 cm	sic	17.8	420.3	127.9	10.5	ND

System 3-73: LR 0.39; DR 0.92; DV 340; UC 3.1; APS 0.2
soil - Kewaunee silt loam, 2-6% slope
PSA 1.20% 1.18% 4.46% 32.04% 45.04% 9.50% 6.48%

Fill	s	12.0	95.4	23,236.5	14.1	2.03
5 cm	sil	28.1	188.4	357.1	22.9	0.59
15 cm	sil	30.6	4,625.7	9,583.1	24.7	3.9
35 cm	sl	29.5	1,073.1	71.5	34.8	ND
55 cm	sic	17.8	420.3	127.9	10.5	ND

System 3-76: LR 0.12; DR 0.65; DV 425; UC 2.5; APS 0.35
soil - Keowns silt loam; partial fill?
PSA 4.99% 3.17% 11.84% 43.10% 30.86% 3.4% 2.73%

Fill	s	5.0	1,164.6	3,039.6	1.54	0.96
5 cm	sil	28.3	117.4	964.1	2.4	0.90
15 cm	sil	49.0	4,321.2	29,600	9.8	8.7
35 cm	sl	34.3	7,022.6	20,700		6.6
55 cm	sl	18.3	5,874.9	11,100		3.5

System 3-82: LR 0.74; DR 0.86; DV 318; UC 3.0; APS 0.40
soil - Plainfield loamy fine sand, 2-6% slope
PSA 10.77% 11.21% 17.00% 41.93% 14.46% 1.27% 3.36%

Fill	s	7.7	6.8	4.9	0.11	0.76
5 cm	s	45.1	17.2	46.5	0.12	2.3
15 cm	s	34.8	28.4	63.1	1.7	15.2
35 cm	s	19.2	56.5	4,643.3	ND	ND
55 cm	s	13.5	4,579.5	3,274.8	ND	ND

System 3-83: LR 0.18; DR 0.16; DV 77.4; UC 1.8; APS 0.16
soil - Solona silt loam, 1-3% slope
PSA 0.77% 0.35% 0.95% 24.09% 68.23% 3.68% 1.96%

Fill	s	7.9	559.1	9,610	4.07	2.7
5 cm	l	20.6	362.5	3,288.5	0.05	1.39
15 cm	l	28.1	410.6	253.2	0.81	6.5
35 cm	sl	16.2	1,027.8	1,107.7	2.6	5.5
55 cm	cl	20.2	12,725.7	9,757.2	7.7	

System 3-110: LR 0.44; DR 0.79; DV 388; UC 9.0; APS 0.15
soil - Sisson very fine sandy loam, 0-2% slope
PSA 0.02% 1.29% 1.47% 35.14% 36.70% 6.02% 21.38%

Fill	ls	13.4	1.1×10^5	18,181	0.01	0.63
5 cm	sl	18.3	303.8	467	0.01	0.13
15 cm	sl	16.0	10,800	2,466	0.02	0.70
35 cm	cl	21.2	1.7×10^5	2,117	0.08	1.08
55 cm	cl	22.1	158.9	1,789.3	0.03	1.12

Table 11.

Bacterial attenuation ratios and moisture contents in the fill and at increasing depths in the soil below 3 rounds of Package 1, 6 rounds of Package 2 and 24 rounds of Package 3 design (averages of all samples analyzed).

Depth	H_2O, %	TC	FC	FS	Ps. a.
Package 1					
Fill	8.5	46	433	58	3.9
5 cm	32.1	126	1,716	0.93	0.28
15 cm	26.0	336	2,870	1.10	0.67
35 cm	22.3	3,033	23,000	0.81	1.06
55 cm	24.8	2,470	7,068	0.43	1.14
Package 2					
Fill	7.9	526	349	3.8	1.33
5 cm	33.0	97	527	0.61	3.8
15 cm	26.8	2,159	1,647	20.5	10.3
35 cm	19.8	3,772	3,863	912	ND
55 cm	17.7	2,748	1,621	816	ND
Package 3					
Fill	9.9	499	88	2.0	0.9
5 cm	24.7	285	105	1.6	0.59
15 cm	24.0	2,228	279	9.8	3.2
35 cm	20.3	3,068	230	188	3.6
55 cm	31.3	4,342	414	25	3.8

Table 12. Bacterial attenuation ratios and moisture contents for fill and soil at increasing depths below all systems studied (averages of all samples from 33 rounds)

Depth	H_2O, %	TC	FC	FS	Ps. a.
All Systems					
Fill	9.3	531	877	4.0	1.2
5 cm	24.8	210	921	1.08	0.76
15 cm	24.8	1,827	2,672	9.3	3.3
35 cm	20.3	3,030	2,878	163	2.33
55 cm	28.3	3,708	3,463	48	2.33

Table 13. Bacterial attenuation ratios (mean of the effluent MPN/100 mls) divided by mean of bacterial counts in the soil (MPN/100 g dry soil) and Cl⁻ (g/g of dry soil) in the fill and at increasing depths in soils below 3 Package 1,6 Package 2, and 24 Package 3 mound designs, and averages for all 33 mound systems.

$$\frac{\bar{X}}{G} = \frac{\text{ratios calculated using arithmetic means of effluent and dry soil values}}{\text{ratios calculated using geometric means of effluent and dry soil values}}$$

Depth	Cl⁻	TC	FC	FS	Ps. a.
Package 1					
Fill	$\frac{21.8}{158}$	$\frac{783}{5,443}$	$\frac{8,864}{51,007}$	$\frac{184}{631}$	$\frac{42}{45.3}$
5 cm	$\frac{2.25}{24.8}$	$\frac{410}{383}$	$\frac{3,900}{5,346}$	$\frac{26.8}{2.8}$	$\frac{1.16}{0.86}$
15 cm	$\frac{6.7}{10.0}$	$\frac{836}{1,284}$	$\frac{11,158}{11,037}$	$\frac{44.3}{4.2}$	$\frac{16.5}{2.5}$
35 cm	$\frac{26.3}{26.0}$	$\frac{3,330}{12,740}$	$\frac{1.9\times10^5}{1\times10^5}$	$\frac{48.5}{3.5}$	$\frac{37.4}{4.5}$
55 cm	$\frac{4.4}{5.7}$	$\frac{29,400}{9,867}$	$\frac{1\times10^5}{28,512}$	$\frac{34.0}{1.73}$	$\frac{37.4}{4.5}$
Package 2					
Fill	$\frac{18.7}{11,957}$	$\frac{2,000}{7,162}$	$\frac{1,147}{4,769}$	$\frac{13.0}{52.3}$	$\frac{19.6}{18.2}$
5 cm	$\frac{3.5}{568}$	$\frac{920}{324}$	$\frac{1,620}{1,755}$	$\frac{47.0}{2.04}$	$\frac{33.9}{12.7}$
15 cm	$\frac{6.2}{24.5}$	$\frac{5,610}{8,638}$	$\frac{7,064}{8,416}$	$\frac{100}{80}$	$\frac{958}{41.5}$
35 cm	$\frac{5.9}{13.1}$	$\frac{54,500}{19,853}$	$\frac{1.7\times10^6}{20,439}$	$\frac{233}{4,802}$	$\frac{ND}{ND}$
55 cm	$\frac{6.4}{16.5}$	$\frac{13,200}{16,166}$	$\frac{11,275}{9,530}$	$\frac{234}{4,803}$	$\frac{ND}{ND}$
Package 3					
Fill	$\frac{21.8}{584}$	$\frac{1,150}{5,854}$	$\frac{1,230}{10,354}$	$\frac{0.93}{23.5}$	$\frac{9.37}{10.5}$
5 cm	$\frac{6.5}{184}$	$\frac{1,260}{1,424}$	$\frac{1,119}{5,241}$	$\frac{22.2}{6.7}$	$\frac{1.36}{2.9}$
15 cm	$\frac{7.4}{29.7}$	$\frac{2,140}{10,610}$	$\frac{2,849}{13,287}$	$\frac{20.4}{46.7}$	$\frac{116}{15.2}$
35 cm	$\frac{5.5}{16.3}$	$\frac{8,150}{16,168}$	$\frac{2,924}{12,129}$	$\frac{6.9}{976.0}$	$\frac{105}{12.3}$
55 cm	$\frac{5.1}{12.0}$	$\frac{7,070}{22,264}$	$\frac{10,281}{21,770}$	$\frac{798}{128}$	$\frac{105}{16.4}$
All Systems*					
Fill	$\frac{18.9}{369}$	$\frac{1,290}{6,527}$	$\frac{1,270}{10,771}$	$\frac{1.2}{49.5}$	$\frac{13.5}{14.7}$
5 cm	$\frac{4.3}{109}$	$\frac{1,110}{926}$	$\frac{1,198}{4,065}$	$\frac{17.5}{4.8}$	$\frac{3.63}{3.36}$
15 cm	$\frac{6.8}{14.5}$	$\frac{2,250}{819}$	$\frac{3,414}{12,038}$	$\frac{22.9}{42.5}$	$\frac{232}{14.9}$
35 cm	$\frac{6.1}{9.4}$	$\frac{9,020}{15,765}$	$\frac{3,642}{14,974}$	$\frac{7.3}{814}$	$\frac{105}{12.3}$
55 cm	$\frac{5.6}{6.7}$	$\frac{7,520}{19,334}$	$\frac{9,400}{17,641}$	$\frac{616}{246}$	$\frac{105}{16.4}$

*The data for Ps. a. for all systems at the 35 and 55 cm depths are averaged from the package 1 and 3 mound system data as no data were obtained from the package 2 mounds at these depths.

Table 14. Bacterial attenuation ratios (mean of the effluent MPN/100 mls) divided by mean of bacterial counts in the soil (MPN/100 ml soil solution) and Cl^- (mg/L of soil solution) in the fill and at increasing depths in soils below 3 Package 1, 6 Package 2, and 24 Package 3 mound designs, and averages for all 33 mound systems.

$$\frac{\bar{X}}{\bar{G}} = \frac{\text{ratios calculated using arithmetic means of effluent and soil solution}}{\text{ratios calculated using geometric means of effluent and soil solution}}$$

Depth	Cl^-	TC	FC	FS	*Ps. a.*
Package 1					
Fill	$\frac{1.83}{11.9}$	$\frac{74.1}{464}$	$\frac{744.6}{4,331}$	$\frac{13.98}{58}$	$\frac{3.57}{3.9}$
5 cm	$\frac{0.72}{7.83}$	$\frac{132}{124}$	$\frac{1,247}{1,716}$	$\frac{8.4}{0.93}$	$\frac{0.52}{9.28}$
15 cm	$\frac{1.74}{2.50}$	$\frac{150}{334}$	$\frac{2,900}{2,870}$	$\frac{11.5}{1.1}$	$\frac{431}{0.67}$
35 cm	$\frac{5.87}{5.72}$	$\frac{7,440}{3,033}$	$\frac{43,480}{23,000}$	$\frac{10.8}{0.81}$	$\frac{8.24}{1.06}$
55 cm	$\frac{1.09}{1.05}$	$\frac{7,300}{2,470}$	$\frac{25,300}{7,068}$	$\frac{8.41}{0.43}$	$\frac{8.99}{1.14}$
Package 2					
Fill	$\frac{1.47}{39.6}$	$\frac{158}{524}$	$\frac{87.4}{349}$	$\frac{1.04}{3.8}$	$\frac{1.56}{1.33}$
5 cm	$\frac{1.16}{46.5}$	$\frac{304}{97}$	$\frac{536}{527}$	$\frac{1.6}{0.61}$	$\frac{11.20}{3.8}$
15 cm	$\frac{1.66}{1.72}$	$\frac{1,500}{2,159}$	$\frac{1,836}{1,647}$	$\frac{25.9}{20.5}$	$\frac{248}{10.3}$
35 cm	$\frac{1.15}{0.70}$	$\frac{10,800}{3,772}$	$\frac{33,472}{3,883}$	$\frac{44.4}{912}$	$\frac{ND}{ND}$
55 cm	$\frac{1.13}{0.78}$	$\frac{2,330}{2,748}$	$\frac{1,998}{1,662}$	$\frac{56.1}{816}$	$\frac{ND}{ND}$
Package 3					
Fill	$\frac{2.16}{31.8}$	$\frac{96.4}{499}$	$\frac{121.8}{88.7}$	$\frac{0.92}{2.0}$	$\frac{0.928}{0.9}$
5 cm	$\frac{1.62}{23.3}$	$\frac{263}{285}$	$\frac{268.8}{105}$	$\frac{4.89}{1.4}$	$\frac{27.7}{0.59}$
15 cm	$\frac{1.77}{3.98}$	$\frac{435}{222}$	$\frac{683.8}{279}$	$\frac{4.89}{9.8}$	$\frac{27.7}{3.2}$
35 cm	$\frac{1.11}{1.98}$	$\frac{1,380}{3,068}$	$\frac{584.7}{230}$	$\frac{1.40}{186}$	$\frac{34.5}{3.59}$
55 cm	$\frac{1.60}{1.50}$	$\frac{1,850}{4,342}$	$\frac{3,182.9}{414}$	$\frac{247.0}{25.0}$	$\frac{53.4}{3.79}$
All Systems*					
Fill	$\frac{1.76}{30.0}$	$\frac{120}{531}$	$\frac{117.9}{877}$	$\frac{0.110}{4}$	$\frac{1.2}{1}$
5 cm	$\frac{1.16}{24.5}$	$\frac{300}{210}$	$\frac{323.5}{921}$	$\frac{4.7}{1}$	$\frac{0.979}{0.76}$
15 cm	$\frac{1.68}{3.22}$	$\frac{556}{1,827}$	$\frac{819.2}{2,672}$	$\frac{5.5}{9.3}$	$\frac{55.5}{3.3}$
35 cm	$\frac{1.22}{1.80}$	$\frac{1,800}{3,030}$	$\frac{728.6}{2,878}$	$\frac{1.46}{162}$	$\frac{21}{2.3}$
55 cm	$\frac{1.58}{1.28}$	$\frac{2,110}{3,708}$	$\frac{2,892}{3,463}$	$\frac{151.5}{48}$	$\frac{29.4}{2.33}$

*The data for *Ps. a.* for all systems at the 35 and 55 cm depths are averaged from the package 1 and 3 mound system data as no data were obtained from the package 2 mounds at these depths.

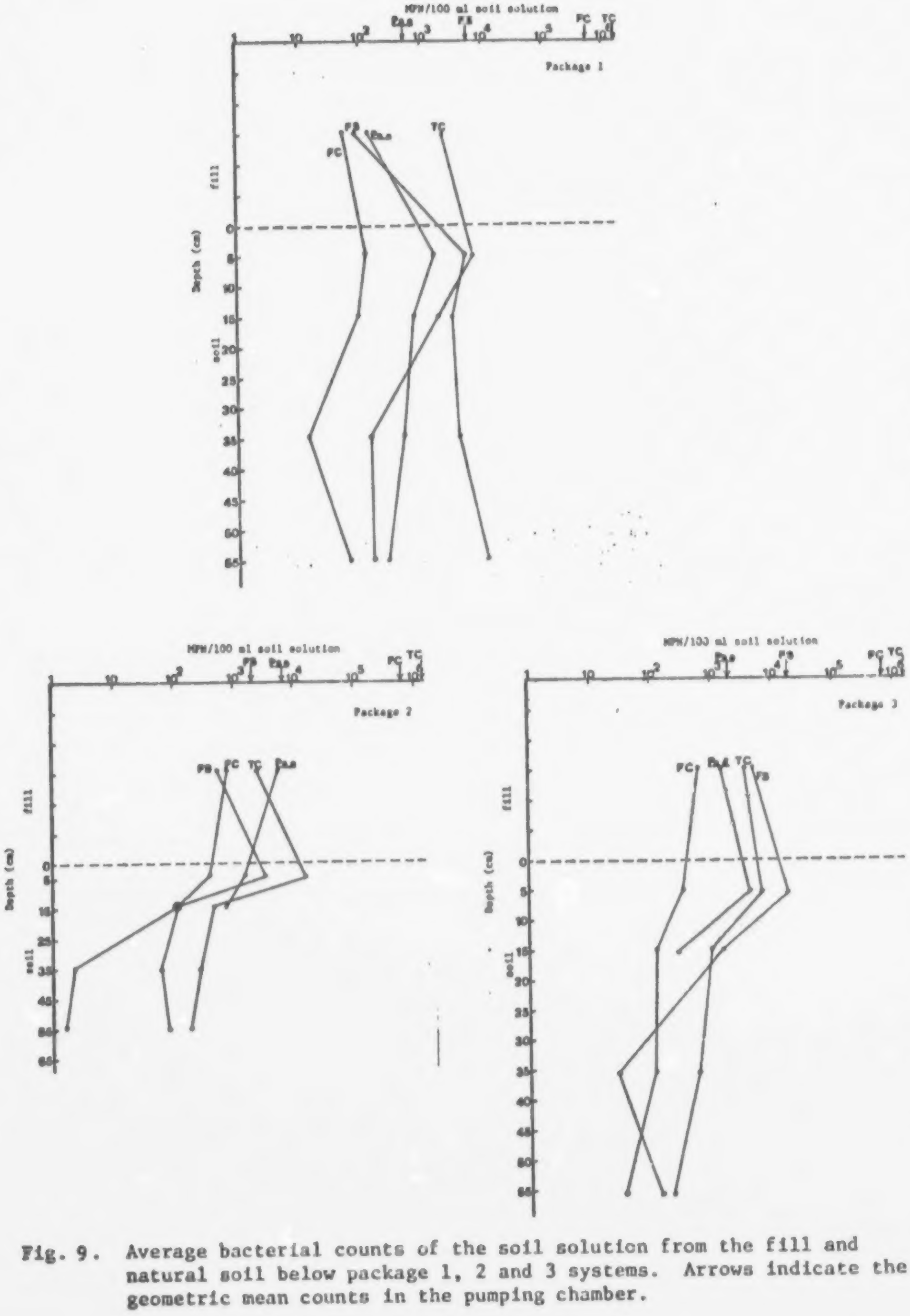

Fig. 9. Average bacterial counts of the soil solution from the fill and natural soil below package 1, 2 and 3 systems. Arrows indicate the geometric mean counts in the pumping chamber.

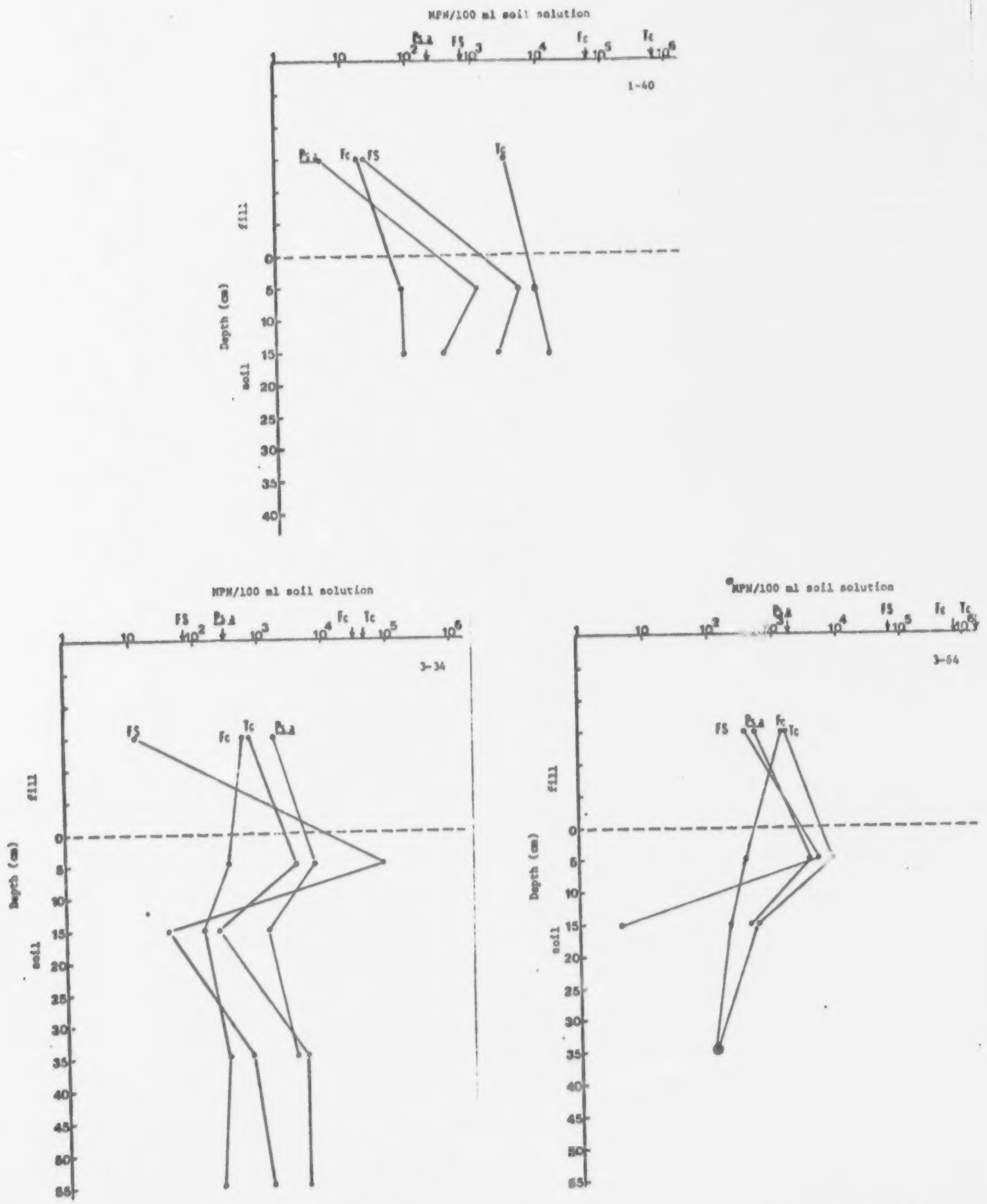

Fig.10. Average bacterial counts of the soil solution from the fill and natural soil under systems 1-40, 3-34 and 3-64. Arrows indicate the geometric mean counts in the pumping chamber.

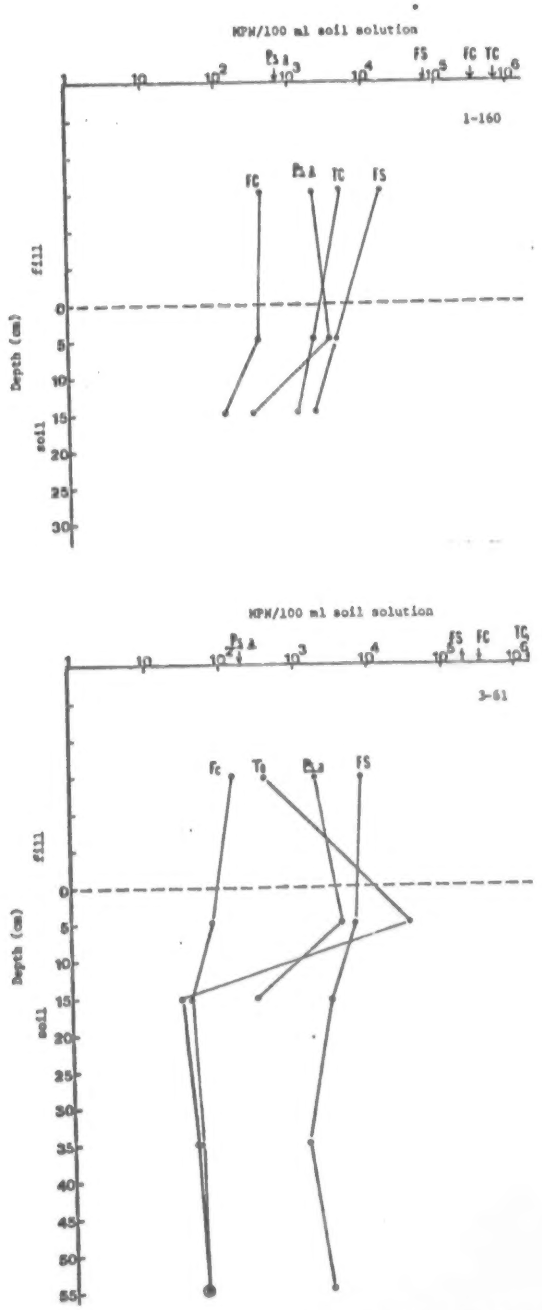

Fig.11. Average bacterial counts of the soil solution from the fill and natural soil under systems 1-160 and 3-61. Arrows indicate the geometric mean counts in the pumping chamber.

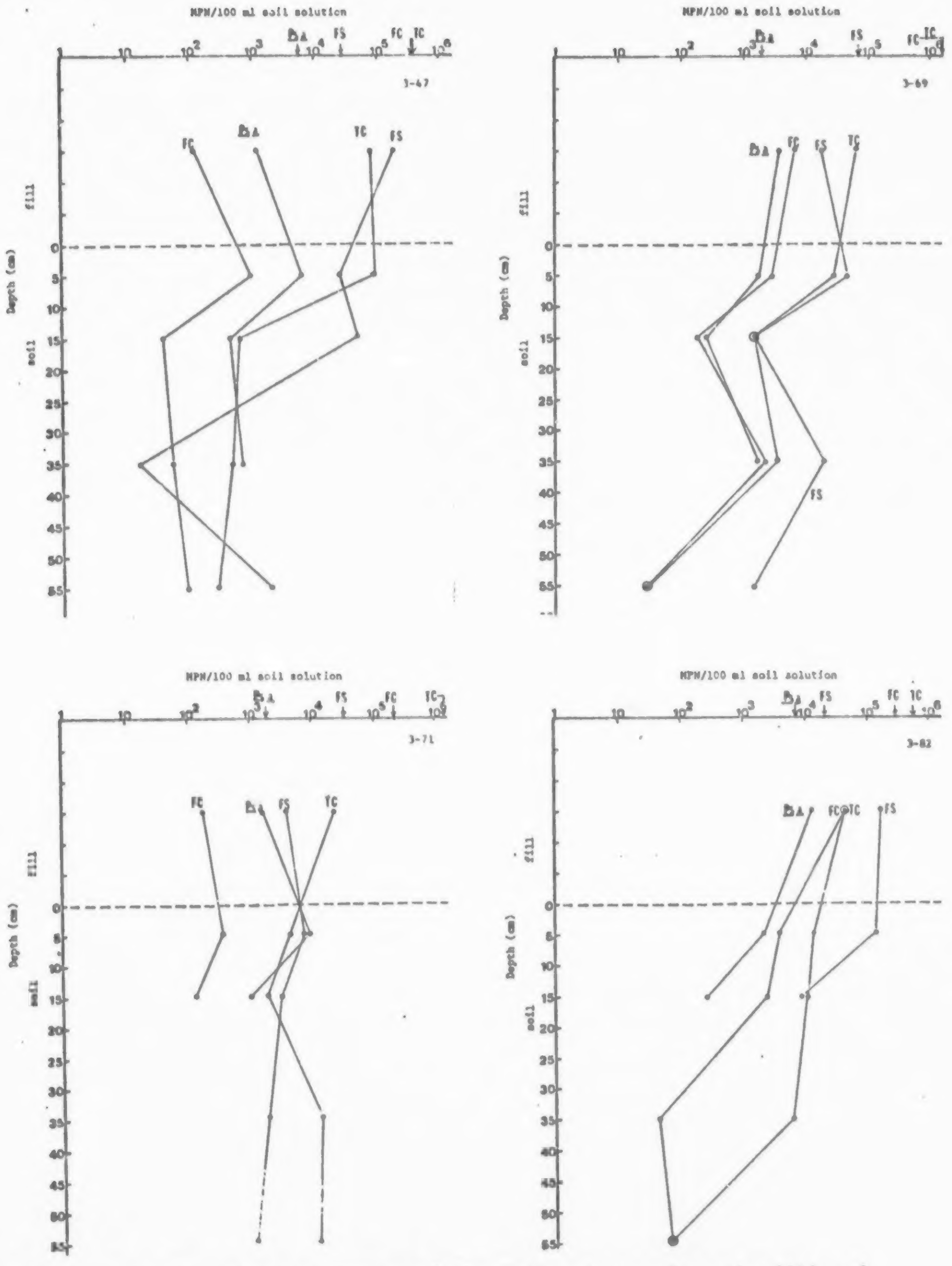

Fig. 12 . Average bacterial counts of the soil solution from the fill and natural soil under systems 3-47, 3-69, 3-71 and 3-82. Arrows indicate the geometric mean counts in the pumping chamber.

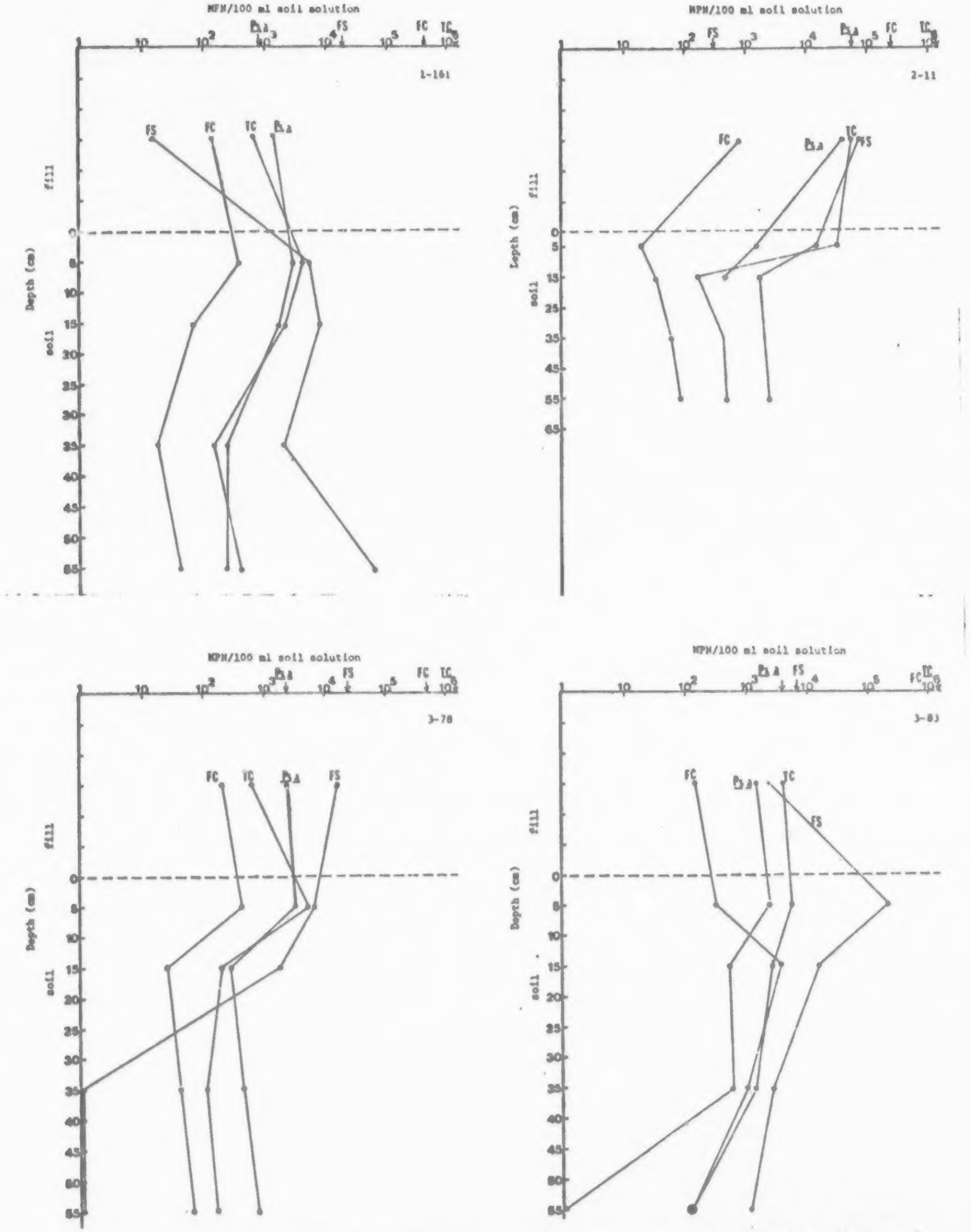

Fig. 13. Average bacterial counts of the soil solution from the fill and natural soil under systems 1-161, 2-11, 3-78 and 3-83. Arrows indicate the geometric mean counts in the pumping chamber.

enter the natural soil. By creating near saturated flow through the sand fill, high dose applications allow bacteria to move with percolating effluent.

The average level of *Ps. a.* at the natural soil surface exceeds the average level of the bacteria in the effluent by 31.4% (cf. Table 12). FS and *Ps. a.* passing through the sand fill are apparently being concentrated at the natural soil surface by physical filtration or are multiplying by aftergrowth after they reach the natural soil. This question cannot be resolved by the data available. FS are enteric bacteria considered to have very limited survival times outside the human body (Clausen et al., 1976); thus the high numbers of FS at the natural soil surface are likely the result of physical filtration. *Ps. a.* appears to be capable of growth in the soil (Ziebell et al., 1975); the data suggest that *Ps. a.* may be growing below the system.

Reasons for the extensive downward movement of FS and *Ps. a.* may be the small size of streptococci (0.6-1.0 m diameter) which allows them to pass through the filtering sand fill, and the mobility (due to polar flagellae) of *Ps. a.* which allows them to travel with the water flow.

For purposes of comparison with bacterial levels in ambient natural soils, counts of TC, FC, FS and *Ps. a.* from surface horizons of soil close to several mound systems and averages derived therefrom are shown in Table 15; similar values were given by Bouma et al. (1972).

Package design differences

By comparing the attenuation ratios shown in Tables 11 - 13, or the bacterial counts at different spots in the mound systems tested, grouped according to the three package designs (Figs. 9-11), it can be

Table 15. Bacterial counts (TC, FC, FS, *Ps. a.*) in samples of natural surface soil (1-5 cm depth) collected near various mound systems, calculated on a soil solution basis (MPN/100 ml soil solution)

System	TC	FC	FS	*Ps. a.*
1-40	5,140	140	2,140	4
1-99	4,100	1,150	3,850	680
1-160	14×10^4	14×10^4	—	—
1-161	1.2×10^4	2,100	1,900	580
2-11	3,800	600	480	960
2-12	5,700	50	2,300	450
2-76	1.3×10^5	1.3×10^5	500	840
2-100	1,500	330	6,600	1,300
2-109	9,520	9,100	3,200	750
2-243	1.1×10^4	95	4,660	930
3-24	1.2×10^4	100	570	1,000
3-47	8,500	85	1,000	1,000
3-54	650	52	260	520
3-61	600	600	260	48
3-64	1.4×10^3	5,200	2,600	520
3-69	1.25×10^4	100	560	1,000
3-71	1.4×10^4	110	6,000	120
3-72	600	48	2,699	529
3-73	1.7×10^4	6,600	720	180
3-74	660	120	7,600	240
3-78	—	—	7,800	160
3-83	1,600	100	660	260
3-110	1,800	1,000	690	240
3-110	2,200	--	--	--
3-112	1,950	980	740	220
3-123	6,000	140	2.0×10^4	250
3-234	970	150	4,100	705
3-273	55,000	725	3,500	600
Arithmetic mean	1.7×10^4	1.1×10^4	3.2×10^3	557
Geometric mean	4.8×10^3	520	1.8×10^3	437
Std. Dev.	3.5×10^4	3.6×10^4	4.1×10^3	339
Std. Error	6.8×10^3	7.1×10^3	808	67
95% Confidence Interval	3.0×10^4 3.6×10^3	2.5×10^4 2.5×10^{-3}	4.8×10^3 1.7×10^3	688 427

concluded that the Package 2 design generally performs a better job of bacterial attenuation than the Package 1 or 3 designs. This is expected as the Package 2 design incorporates 2 ft of sand fill over the natural soil, versus 1 ft in the Package 1 and 3 designs.

The bacterial attenuation ratios show that no design has fill which adequately reduces FS or *Ps. a.* numbers before effluent reaches the natural soil. This phenomenon is the result of a flushing action due to quasi-saturated flow through the sand fill following dosing events. Saturated flow is a function of 2 variables, namely, dosing rate and texture of the sand fill. The dominant factor is the dosing rate which averaged 0.74 gal/ft^2/dose (range 0.14-1.05 gal/ft^2/dose) for all systems studied. A 0.74 gal/ft^2/dose dosing rate corresponds to a flow of about 77 cm/day (see Table 7) which is above the maximum flow of 40 cm/day recommended by Bouma et al. (1972) for optimum bacterial attenuation in a medium sand. However the sand fill being used in several mound systems is of a finer texture than the sand fill recommended for use in mound systems. The uniformity coefficient for the sand used by Bouma et al. (1972) was 1.8 with an average particle size >0.25 mm. The average uniformity coefficient for the mounds being monitored was 5.7 (range 1.8-100+, SD 6.7), with an average particle size of 0.15 mm (range 0.10-0.60 mm).

The less uniform and finer, textured fill enhanced bacteria attenuation in some mounds despite high dosing rates. This is undoubtedly due to the better straining action associated with the more diverse and finer particle sizes in some fills (cf. Butler et al., 1954). Comparison of the ability of selected individual systems to remove bacteria follows together with a discussion of the relationships to the dosing rate and the texture of the sand fill.

Individual systems

Four different situations are of interest when considering the bacterial travel through individual mound systems:

1. Mounds with a high dosing rate, and fill of high uniformity coefficient;
2. Mounds with a low dosing rate, and fill of high uniformity coefficient;
3. Mounds with a high dosing rate, and fill of low uniformity coefficient;
4. Mounds with a low dosing rate, and fill of low uniformity coefficient.

Situation 4 is the best for mound operation; a low dosing rate combined with a uniform medium-textured sand should maintain unsaturated flow within the mound system and prevent bacterial travel without danger of biological clogging.

A multiple regression was performed regarding the bacterial attenuation ratios for TC and FC, the uniformity coefficient of the sand fill, and the dosing rate. The multiple correlation coefficient for TC travel through the fill was 0.79 (n=20) and for FC travel was 0.82 (n=20); both values fall in the 99.5% confidence limits. The respective regression equations, where Y = bacterial attenuation ratio (i.e. geometric mean of TC or FC in effluent: geometric mean of TC or FC in 100 ml of soil solution), X_1 = dosing rate (gal/ft^2/dose), and X_2 = uniformity coefficient of the sand fill, are:

For TC (multiple regression coefficient = 0.7933):

$$Y = 426.6 - 223.45\ (X_1) + 24.5\ (X_2)$$

$$r\ X_2X_1 = -0.0525$$

$$r\ X_1Y = -0.1283$$

$$r\ X_2Y = 0.7885$$

For FC (multiple regression coefficient = 0.8250):

$$Y = 887.8 - 591.9\ (X_1) + 105.9\ (X_2)$$

$$r\ X_2X_1 = -0.0525$$

$$r\ X_1Y = -0.0990$$

$$r\ X_2Y = 0.8231$$

By using the recommended uniformity coefficient (1.8) for the medium sand fill (Bouma et al., 1972) the equations indicate that a dosing rate of about 1.2 gal/ft^2/dose will result in only 99.5% removal of TC, but 99.7% removal of FC. However, a dosing rate of 0.3 gal/ft^2/dose affords a 99.75% removal of TC and 99.99% removal of FC. Regression equations for the removal of FS and *Ps. a.* did not give good correlations, probably because these bacteria are either being concentrated or regrowing at the natural soil surface. However, the dosing rates for the various fill materials would be much less for the same percentage removal of FS and *Ps. a.* Examples of how dosing rate and fill consistency affect bacterial travel in individual systems are discussed below. The mounds considered were grouped according to the four dosing rate/uniformity combinations indicated above (p. 89; see Table 16). Figures 10-13 depict the average bacterial levels found at increasing depths below these systems, indicating the extent of bacterial travel, as well as the average bacterial levels found in the

Table 16. Hydraulic loading and fill characteristics of selected individual mound systems, grouped according to important dosing rate/uniformity coefficient combinations (cf. text, p. 88).

	Individual Mound Systems			
Systems	Dosing rate ($gal/ft^2/dose$)	Loading rate ($gal/day/ft^2$)	Uniformity coefficient of fill	% Fines of fill (silt and clay)
Group 1				
1-40	0.89	0.51	30.9	4.8
3-34	0.97	0.05	7.5	5.5
3-64	1.05	1.16	9.9	15.7
Group 2				
1-160	0.12	0.25	16.7	8.5
3-61	0.14	0.23	5.4	3.5
Group 3				
3-47	0.95	0.21	3.7	1.8
3-69	0.96	0.37	3.2	2.5
3-71	1.02	0.25	2.0	3.3
3-82	0.86	0.74	3.0	3.0
Group 4				
1-161	0.58	0.22	2.2	9.5
2-11	0.75	0.35	2.2	3.0
3-78	0.65	0.12	2.5	2.4
3-83	0.16	0.18	1.8	2.3

effluent from each system.

The 3 Group 1 mounds have each a high dosing rate and a high uniformity coefficient (Table 16). Whenever sampled, mound 3-64 exhibited ponding at the sand fill/gravel interface, or saturated conditions within the fill. Neither mound 1-40 nor 3-34 exhibited ponding or saturated conditions at any time.

By comparing the bacterial counts obtained from the soil surface, it can be seen that for efficiency in bacterial attenuation 3-34 is the worst, 1-40 is better, and 3-64 is best. In mound 3-34 the flow immediately after each sporadic dosing event (1 dose every 20 days) is saturated and the uniformity of the sand does not allow it to function as a good filtering medium. System 1-40 has a lower dosing rate and a higher uniformity coefficient, making it more effective in removing bacteria from the percolating effluent. System 3-64 has a high dosing rate, but also a high uniformity coefficient and reduces bacteria levels best within this group. Thus, less uniform fill improves bacterial reduction at high dosing rates, but may ultimately introduce other complications (crusting, ponding, hydraulic failure) that offset this benefit.

System 3-64 exhibited a ponded condition in the absorption bed four times during the monitoring program, suggesting that it may have developed a crust at the gravel/fill interface. Ziebell et al. (1975) showed that bacterial travel in a ponded crusted situation was less than that in an unponded uncrusted condition; system 3-64 may achieve greater bacterial attenuation due to a crusted state.

All three systems in Group 1 are operating outside the recommendations given for mound construction, yet none has failed and the worst system, 3-34, still accomplished a 92% reduction in TC before the percolating effluent encounters the natural soil surface.

The Group 2 mounds (Table 16) both have low dosing rates, dose approximately twice each day, and have relatively high uniformity coefficients for the sand fill. Since these systems should be operating in an unsaturated state, bacterial movement can be attributed to the transient saturated conditions following each dosing event caused by the low uniformity of the fill. A saturated zone at the soil/fill interface was observed in system 1-160 15 minutes after a dosing event. System 1-160 is located over a Morley silt loam with a percolation rate of 100 min/in, so the saturated condition here was probably only temporary (Fig. 11).

System 3-61 is more efficient in removing nonmotile bacteria (TC,FC,FS) than system 1-160 because of the lower uniformity coefficient of its sand fill. This allows for a quicker return to an unsaturated situation in the fill after a dosing event, and lowers the travel of nonmotile bacteria.

The reverse is true for motile bacteria (*Ps. a*), which travel with the water flow. The less dense the filter medium, the more direct the movement of these bacteria. The more uniform fill in system 3-61 allows more travel of *Ps. a*. than the less uniform fill of system 1-160.

Comparison of the values from Group 1 and Group 2 systems shows that lowering the dosing rate increases the removal of TC. Ongoing work on bacterial travel through various textures of filtering media at different dosing levels may allow dosing rates to be adjusted to the fill material texture to maximize bacterial attenuation (Harkin and Iskandar, 1979).

Group 3 mounds meet the recommended uniformity requirement for sand fill but are being dosed at a level much higher than recommended (Table 16). The surging effect of each dose and the ease of travel through the uniform sand fill make this group of systems the least effective in bacterial removal with differences governed by the dosing rate and total load. The least effective system is mound 3-82 (Fig. 12).

System 3-82 has the lowest dosing rate but the highest loading rate of the 4 Group 3 systems. On the average, system 3-82 doses once every 1.1 days, while system 3-69 doses once every 2.6 days, 3-71 once every 4.1 days, and 3-47 once every 4.5 days. Thus bacteria travel is easier through the sand fill in system 3-82 than in the other three systems. Similar reasoning explains why system 3-69 allows more TC and FC movement to the natural soil than system 3-71. System 3-47 should be the most effective at bacterial attenuation but is in fact the second least effective, because it is used primarily on the weekends as a vacation home. Thus this system was dosed more often on weekends and actually had intermittent loads higher than either systems 3-69 or system 3-71.

Group 4 mounds are systems operating within or close to the recommended guidelines for mound construction, receiving low or relatively low dosing rates and containing uniform medium sand fill (Table 16). These systems perform best in bacterial attenuation and exemplify how mound systems should work.

The best system is 3-78 which shows a 99.9% reduction in TC at the soil surface. No system studied was able to remove FS or *Ps. a.* to any great extent, but systems 3-78 and 3-83 prevented their movement deep into the natural soil. Here saturated flow at the natural soil surface obviously does not occur with each dose. These mounds are operating in an unsaturated state in the sand fill, preventing saturated flow from a dosing event from reaching the natural soil surface (Fig. 13).

Factors affecting bacterial travel

Bacterial travel into and through the natural soil is affected mainly by the texture of the soil, and by the water content in the soil profile. Coarse-textured soil, such as a sand or loamy sand, allows greater bacterial travel than finer-textured soils such as a silt loam or a clay loam. Systems 3-34 (over loamy sand and sandy loam), 3-47 (sandy loam), 3-71 (sandy loam), and 3-82 (sand) allow greater TC penetration into the natural soil than systems located over fine-textured soils.

In addition, the bacterial counts increase whenever any textural change or an area of saturation or groundwater occurs in the soil profile. Examples of this phenomenon (cf. Table 10) can be seen in systems 1-161 (saturated zone at 45 cm), 3-34 (coarser to finer texture change at 25 cm, LS-SL), 3-71 (saturated zone at 30 cm), and 3-73 (coarser to finer texture change at 25 cm, SiL-SiC). This effect confirms that bacteria are traveling with the water percolating through the system. When percolating effluent encounters a saturated zone, the water movement stops or slows down and allows the bacteria to concentrate in the soil at that level.

Changes in texture in the soil profile produce two effects: first, finer-textured soil has a better filtering action upon bacteria and may trap them on the surface of a finer-textured horizon (Butler et al., 1954); second, if the soil becomes saturated due to finer texture, bacterial travel may be increased due to saturated flow within the finer-textured horizons. An example of the first situation can be seen in system 3-69 where the bacteria are trapped on the surface of the transition zone between a silt loam and silty clay. The bacteria do not travel further into the profile because the system is operating in an unsaturated condition. The second situation is encountered in systems 3-34, 3-71, and 3-73, where increases in bacterial numbers occur at the level of textural change but the movement of bacteria continues down into the profile because of saturated flow conditions.

In summary, all of the mound systems performed well in removing coliform bacteria, but several mounds allowed movement of fecal streptococci to the natural soil because the high dose volumes are promoting quasi-saturated flow through the fill. While most of the systems allowed for enrichment of *Ps. a.* at the natural soil level, evidence indicates (Ziebell et al., 1974) that this is the result of aftergrowth and not failure of the system to remove bacteria from percolating effluent.

Nitrogen

All mounds

The forms of nitrogen (N) examined were ammonium nitrogen (NH_4-N), nitrate nitrogen (NO_3-N) and total Kjeldahl nitrogen (TKN). These were related to the chloride (Cl^-) concentrations, since Cl^- is a discrete ion that does not change during passage through soil. All Cl^- and NO_3-N values are calculated on a soil solution basis, and the NH_4-N and TKN nitrogen on a dry weight of soil basis, since these are the media with which these chemical species are associated.

In general, the mound systems allow nitrification within the sand fill, denitrification at the natural soil surface, and either nitrification or continued denitrification in the natural soil depending upon the moisture level and texture of the soil. The immobilization of NH_4-N in bacterial tissue varies from system to system with the moisture characteristics. In general NH_4-N is much higher through the treatment depth than in conventional systems due to the dosing network, which tends to force ammonium through the sand fill into the natural soil at the high dosing rates currently being used.

Figure 14 plots the average values of TKN, NH_4-N, NO_3-N, and Cl^- versus depth of all systems studied. Since NO_3-N is formed by nitrification in the sand fill, in general the systems must be aerobic within the sand fill. The level of NO_3-N increases to the sand/soil interface and continues 5 cm into the natural soil. It then drops due to either denitrification or immobilization of the nitrate in bacterial cells.

Comparison of the NO_3-N and Cl^- levels in the soil shows that denitrification starts at the 5 cm depth and continues to the 35 cm depth into the soil. Between 35 and 55 cm, no absolute change in NO_3-N level is indicated, however comparison with the Cl^- level reveals that further denitrification is occurring.

TKN and NH_4-N show similar trends to NO_3-N (lower levels within the

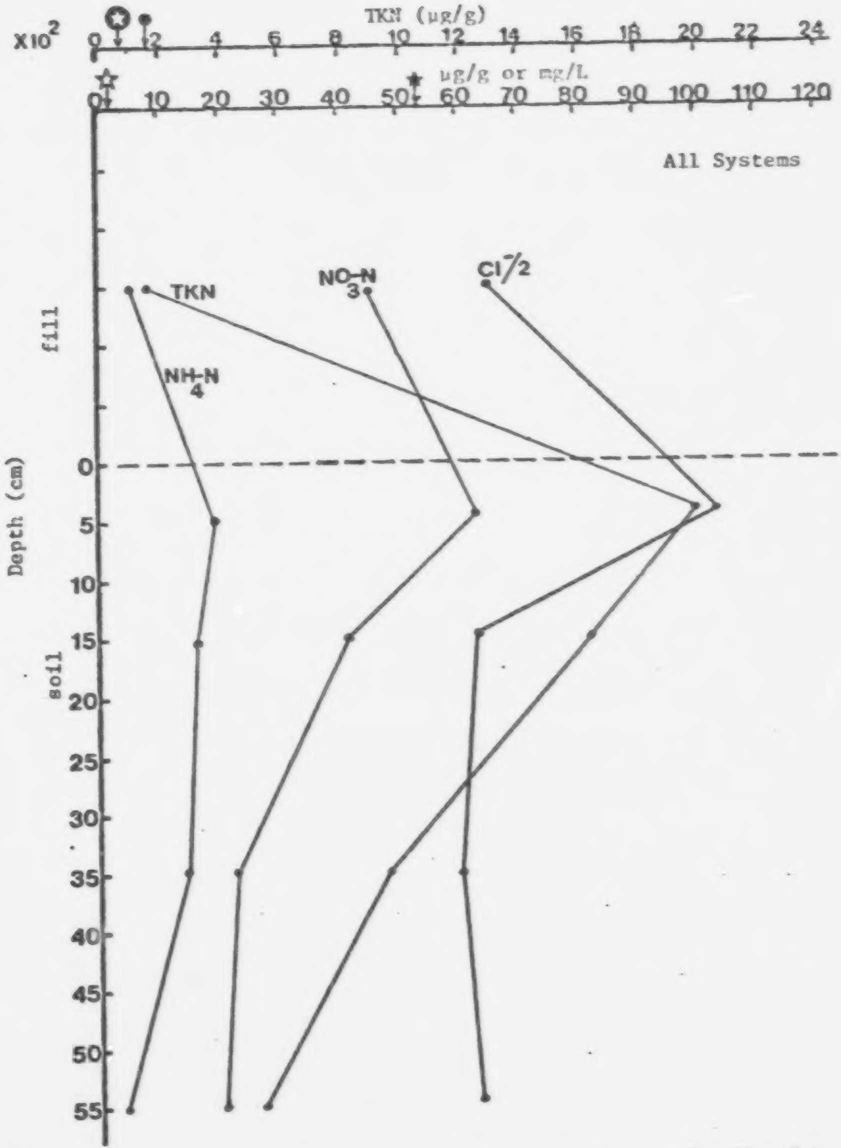

Fig. 14 . Average TKN, HN_4-N, NO_3-N, Cl^- in the fill and natural soil under 33 mounds.

✪ = TKN, ★ = NH_4-N, ☆ = NO_3-N, ● = Cl^-

in the pumping chamber.

sand fill with concentration occurring 5 cm into the natural soil). Below the fill/soil interface, both TKN and NH_4-N decrease with depth and approach back-ground levels in natural soils at a 55 cm depth. For purposes of comparison, levels of the various N forms in samples of ambient surface soils sampled close to several mounds are shown in Table 17.

Two factors differentiate mounds from conventional soil absorption fields: denitrification of NO_3-N and the extent of NH_4-N travel into the soil. In a conventional system (Walker et al., 1973), when septic tank effluent passes beyond the clogging zone there is a build up of NO_3-N with little or no denitrification and a sharp reduction of NH_4-N due to immobilization by bacteria within and just below the clogging mat. Nitrogen transformations in a conventional soil absorption field are governed by the extent of the clogging mat and the unsaturated zone below the crust that allows formation of NO_3-N.

Thanks to the dosing system, mound systems do not have a clogging mat, so that N transformations are governed by the moisture content of the system, which determines whether the system is operating aerobically (promoting nitrification) or anaerobically (promoting denitrification), and by the amount of organic matter in the natural soil plus any fresh organic carbon traveling with the effluent to the soil surface. Readily available biodegradable organic matter is necessary for denitrification (Sikora and Keeney, 1975).

NO_3-N is readily formed from NH_4-N in the sand fill; all but one of the systems studied maintained unsaturated--and therefore aerobic--conditions within the fill except for a short period immediately

Table 17. TKN and NH_4-N in $\mu g/g$ of dry soil, and NO_3-N in mg/L of soil solution, for soil samples collected at a 5 cm depth from soils surrounding several mound systems.

System	TKN	NH_4-N	NO_3-N
1-40	235.5	8.21	1.64
1-99	387.6	3.41	4.30
1-160	383.7	7.14	1.97
1-161	161.3	3.77	4.63
2-11	445.01	4.31	1.62
2-12	307.7	0.0	6.35
2-76	661.5	3.3	0.98
2-105	547.1	5.5	0.46
2-109	119.8	11.07	0.98
2-243	457.2	3.0	0.53
3-24	782.1	10.6	31.4
3-47	374.0	3.0	0.35
3-54	889.6	1.75	0.44
3-61	52.6	4.5	2.1
3-64	945.9	2.7	0.46
3-69	682.0	1.9	0.21
3-71	--	17.6	3.3
3-72	--	6.25	7.21
3-73	83.9	1.5	10.25
3-74	377.9	0.0	7.02
3-78	697.1	1.4	3.2
3-83	116.0	0.42	0.28
3-110a	--	0.0	7.89
3-110b	135.8	2.66	5.70
3-112	--	2.67	5.6
3-123	111.5	3.22	0.0
3-234	707.7	2.16	0.65
3-273	171.8	3.79	1.68
n	24	28	28
Arithmetic mean	405.2	4.1	3.88
Geometric mean	306.3	0.69	1.23
Std. Deviation	274.2	3.9	6.07
Standard Error	56.0	0.74	1.15
95% Confidence	514.8	5.6	6.10
Interval	295.5	2.7	1.60

following each dosing event. This situation promotes extensive nitrification in the sand fill. On the average 43% of the nitrate formed in the sand fill and soil is denitrified as it passes through the surface horizons and through the natural soil because the natural soil surface is generally anaerobic (saturated). Thus, in general, mounds reduce the level of NO_3-N leaving the system more effectively than conventional systems. Assuming that no further N transformations occur below the 55 cm depth, the average NO_3-N flux should be 19.5 mg/L into the groundwater (Fig. 16).

Walker et al. (1973b) calculated the N budget for a conventional system and concluded that the NO_3-N input into groundwater will equal naturally occurring NO_3-N flux if one system is located in every 6 acres.

However, only 4 systems were analyzed in this study and N inputs of 8.2 gr N/person/day, all of which was assumed to be converted to NO_3-N, and water use of 1,136 L/day were used to derive these figures. For the 33 mounds monitored and the 19 homes that gave reliable flow data, the N inputs were only 4.93 kg N/person/yr and the water use was only 658 L/day. Not all of the septic tank effluent N was converted to NO_3-N; about half remained as organic N in the fill or soil of the seepage area. Thus, using the N and flow data measured for the mounds, a corresponding figure can now be derived for a permissible density of mound systems.

Keeney and Gardner (1975) calculated that the N flux required to bring a soil percolate to the 10 mg/L critical NO_3-N level is 2.27 lb N per acre-inch. Assuming that natural inputs of NO_3-N from precipitation

(5 lb/acre/yr) and organic matter decomposition (5 lb/acre/yr) (Keeney and Gardner, 1975) are added to the measured inputs from an average mound system (19.5 mg/L), it is possible to calculate the density of mound systems per acre that would allow the percolate to reach the 10 mg/L critical level.

Nitrogen Inputs:

Percolate from mounds
(19.5 mg/L x 658 L/day x 365 days) = 10.11 lb N/yr

Nitrogen from precipitation = 5.0 lb N/acre/yr

Nitrogen from organic matter
decomposition = 5.0 lb N/acre/yr
Total N inputs 20.30 lb NO_3-N/acre/yr

Water Budget:

Water flux from mound systems
(658 L/day x 365 days
102,778 L/acre-in) = 2.34 acre in/yr

Infiltration from precipitation
(W.R.M. Workshop, 1979) = 9.6 acre in/yr
Total deep percolation = 11.94 acre in/yr

20.30 lb N/acre/yr 11.94 acre in/yr = 1.70 lb N/acre/yr

$$\text{Critical density} = \frac{1.70}{2.27} = 0.75 \text{ acre}$$

∴ 1 mound system per 0.75 acre would allow the percolate to reach the 10 mg/L critical level.

These calculations assume that there is no groundwater flux below the mound systems, and therefore give an underestimate of the actual density of mound systems allowable per acre. Conventional septic systems at current densities seldom cause well water NO_3-N levels above 10 mg/L (Delfino, 1977). However, because rural developments often have small individual lots, care should still be taken to locate water supplies as far from any waste disposal system as possible considering lot limitations. Moreover, as is discussed later, the mound systems have not reached a steady state equilibrium and thus the N flux figures may change as the systems age. Realistically, it appears that 2-4 mound systems could be tolerated per acre without causing excessive well water nitrate levels, since dilution of mound seepage with groundwater of lower NO_3-N content normally occurs. Measurements of the NO_3-N in groundwater close to mound systems (p. 154) did not reveal signs of significant nitrate enrichment, so that a mound density higher than the 1.3 per acre indicated by the above calculation is probably safe in Wisconsin.

Package design differences

Figure 15 depicts the average N transformations occurring below mounds of the 3 package designs. The major differences appear to be in the treatment of NH_4-N by the Package 1 type mounds and the N transformations occurring in the Package 1 and 2 systems.

Package 1 systems incorporate a trench configuration over slowly permeable soils; Package 2 and 3 have absorption beds. The trench design increases the treatment area by adding the areas of the trench sidewalls to the treatment zone. Thus more NH_4-N should be immobilized in the sand fill for Package 1 mounds than in Package 2 and 3 mounds. In practice, Package 1 mounds do indeed prevent NH_4-N from reaching the natural soil better than Package 2 and 3 mounds.

Package 1 mounds also permit more nitrification in the fill. This is important as a prelude to denitrification. Package 1 designs nitrify the effluent up to levels of 93 mg/L while Package 2 and 3 designs allow for average NO_3-N accumulations of only 41.5 and 47.5 mg/L at the

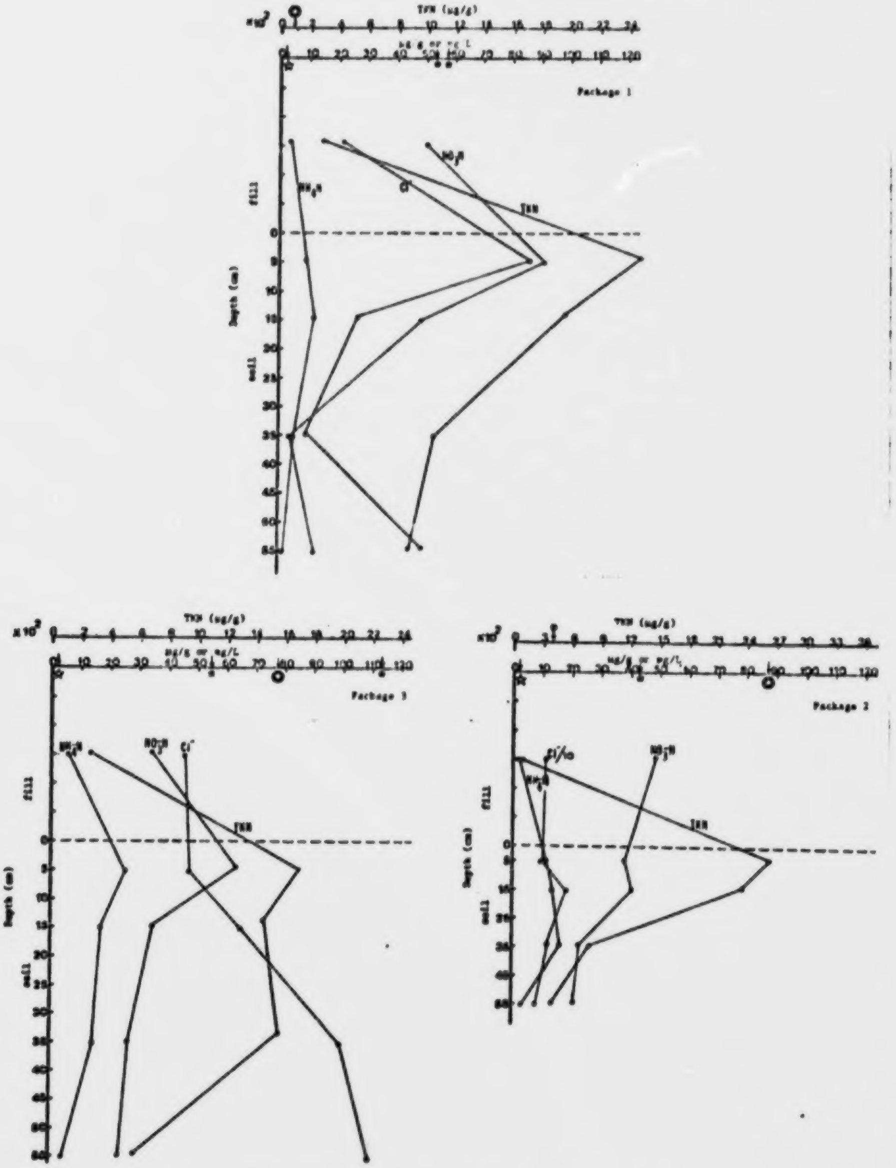

Fig. 15. Average TKN, NH_4-N, NO_3-N and Cl^- in the fill and natural soil under Package 1, 2 and 3 mounds.

○ = TKN, ★ = NH_4-N, ☆ = NO_3-N, ● = Cl^- in the pumping chamber.

natural soil surface. Package 1 mounds thus make more NO_3-N available for denitrification in the underlying finer-textured, less permeable soils.

The sites of N transformations in Package 2 designs differ from those in the other designs. All three designs show nitrification in the sand fill, however only Package 2 designs undergo loss of NO_3-N before the 5 cm depth in the natural soil; Package 1 and 3 designs do not experience NO_3-N removal until after 5 cm into the natural soil. In addition, only Package 2 designs show nitrification between 5 and 15 cm in depth.

Package 2 designs are for areas with permeable soils over shallow creviced or porous bedrock. All the Package 2 mounds studied are over loam soils except for system 2-12 which is over a sandy soil. Because Package 2 mounds have 2 feet of sand fill, some NO_3-N is probably being reduced by immmobilization at the natural soil surface, with slight nitrification continuing at the 5-15 cm depth, followed by denitrification beginning at the 15 cm depth and continuing for the distance monitored. This suggests that, under present dosing practices, Package 2 designs maintain an unsaturated zone to a depth of 15 cm into the natural soil as opposed to only 5 cm for systems of Packages 1 and 3.

Nitrogen transformations in individual systems

The discussion of N transformations for individual systems follows the format described for bacterial attenuation (see p. 88).

Among Group 1 mound (Table 16), systems 3-34 and 3-64 have high dosing rates and high uniformity coefficients but differ in the frequency of dosing: system 3-34 doses once every week and system 3-64 once every day. The resultant difference in N transformations is readily seen in Fig. 16 and can be attributed to the constant quasi-saturated condition in 3-64 and the prolonged unsaturated state of 3-34 between doses.

By operating in a more saturated, anaerobic state, system 3-64 does not allow nitrification of the effluent NH_4-N, so there is little nitrate formed under the mound. In addition, the anaerobic state slows the immobilization of NH_4-N and allows extensive transport of NH_4-N into the soil profile. This indicates that constant saturation slows the decomposition of organic material and stops nitrification while the cycle of aerobic/anaerobic conditions promotes the decomposition of organic material and increases nitrification and denitrification. Similar observations have been made for paddy soil model systems (Patrick and Reddy, 1974, 1976).

System 3-34 doses at a level that produces quasi-saturated flow following a dosing event but the extended time between doses allows the system to return to an aerobic state betwen doses, promoting nitrification. Denitrification in system 3-34 begins at the 15 cm depth into the soil and continues to a depth of 35 cm, when nitrification again occurs. The denitrification in system 3-34 is surprising. Soil moisture data (Table 10)show that the system operates in an unsaturated condition throughout the soil profile, so the oxygen demand of the material deposited at the soil surface must be preventing diffusion of oxygen into the region 15-35 cm in depth below the soil surface. Thus, this area, although unsaturated, is operating anaerobically and

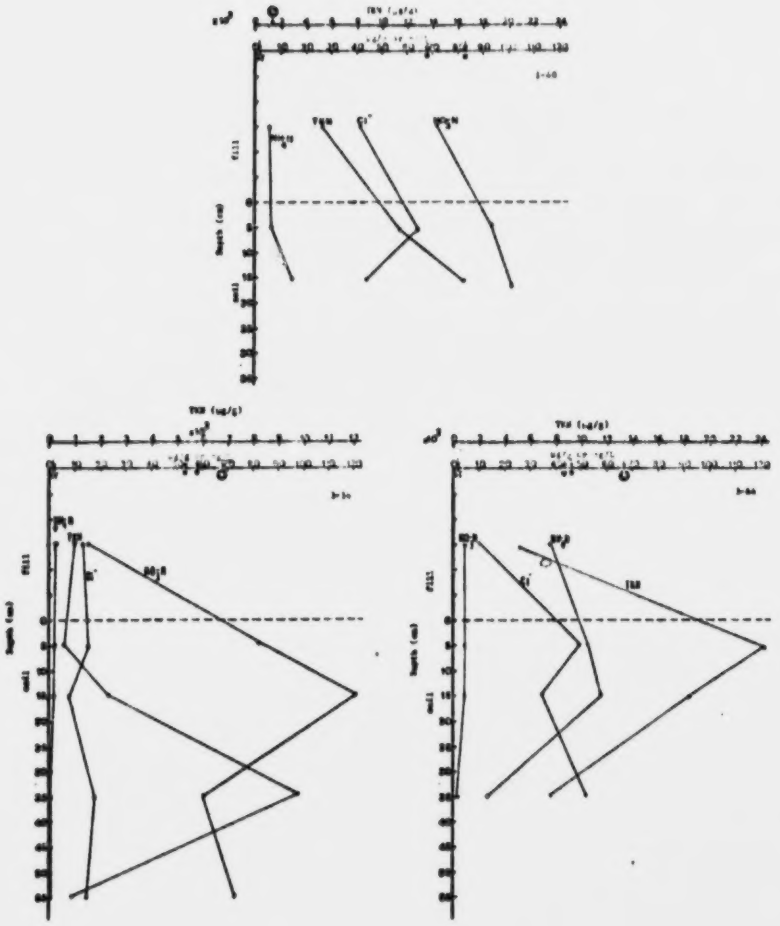

Fig. 16. Average TKN, NH_4-N, NO_3-N and Cl^- in the fill and natural soil under systems 1-40, 3-34 and 3-64.

⊙ = TKN, ★ = NH_4-N, ☆ = NO_3-N, ● = Cl^- in the pumping chamber.

denitrifying the percolating effluent. At the 35 cm depth the oxygen from the surrounding soil is entering the soil under the mound and allowing for nitrification of the percolating effluent.

Both systems differ markedly in the values obtained for TKN. System 3-64, which is usually in a saturated state, is depositing the bulk of the N at the soil surface, while system 3-34, which is usually unsaturated, is depositing it 35 cm into the soil. This difference in behavior can also be attributed to the different soils underlying each system. System 3-64 is located over a Morley silt loam, a fine-textured soil that does not permit the high dose volumes to "push" organic N compounds into the soil. System 3-34 is located over Boyer loamy sand, a coarse-textured soil, which allows the high dose volumes to move the organic N compounds into the soil profile to a depth of 35 cm.

Of the Group 2 systems, which have low dosing rates and high uniformity coefficients (Table 16), system 1-160 operates in a saturated or nearly saturated state, indicated by the low level of nitrate formation. In addition, NH_4-N is quickly immobilized even under the saturated conditions due to the high dosing frequency with small amounts of effluent, whose NH_4-N can be effectively immobilized in the trench bed design between doses. The low dosing rate and the high uniformity coefficient of the sand fill prevent leaching of NH_4-N into the natural soil, as indicated by the TKN peak at the natural soil surface (Fig. 17).

System 3-61 differs from system 1-160 in two respects: first, it has a bed rather than a trench absorption field, second, the uniformity coefficient of its sand fill is much lower. These differences result in

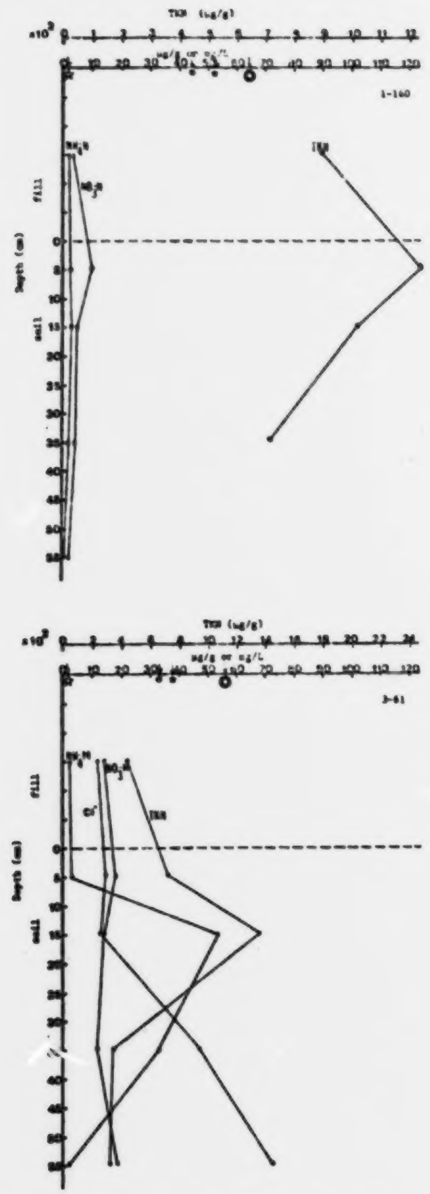

Fig. 17. Average TKN, NH$_4$-N, NO$_3$-N and Cl$^-$ in the fill and natural soil under systems 1-160 and 3-61.

○ = TKN, ★ = NH$_4$-N, ☆ = NO$_3$-N, ● = Cl$^-$ in the pumping chamber.

more nitrification in the sand fill (14 v. 1.86 mg/L NO_3-N) and less immobilization of NH_4-N (maximum level 15 vs. 5 cm) into the soil. As in system 1-160, the TKN maximum coincides with the maximum NH_4-N (at 15 cm for 3-61 vs. 5 cm for 1-160).

Group 3 mounds (Table 16) have all high dosing rates and low uniformity coefficients. System 3-82 shows evidence of extensive denitrification of the effluent as it passes into the natural soil and enrichment of NH_4-N at the natural soil surface (Fig. 18). This system denitrifies 86% of the nitrate formed within the sand fill, due to the high water content (Table 9) of the natural soil (Plainfield loamy fine sand), which indicates that the soil is frequently saturated and anaerobic. At the 35 cm depth, the system returns to an aerobic state, but only slight nitrification occurs after the 35 cm depth (Fig. 18).

System 3-82 also shows enrichment of NH_4-N in the soil. Again the high dosing rate is transporting NH_4-N in effluent through the sand fill into the natural soil. The anaerobic state of system 3-82's soil is shown by the high NH_4-N concentration. The NH_4-N concentration remains high until after the 35 cm depth where the aerobic condition promotes immobilization and nitrification.

System 3-82 shows a TKN peak at 5 cm depth in the natural soil, indicating that organic N compounds remain relatively stable in the anaerobic zone. The TKN follows the NH_4-N concentration and falls only after the effluent enters the aerobic zone beyond the 35 cm depth.

System 3-69 differs from system 3-82 in two respects: the natural soil returns to an aerobic state between the 5 cm and 15 cm depths, and

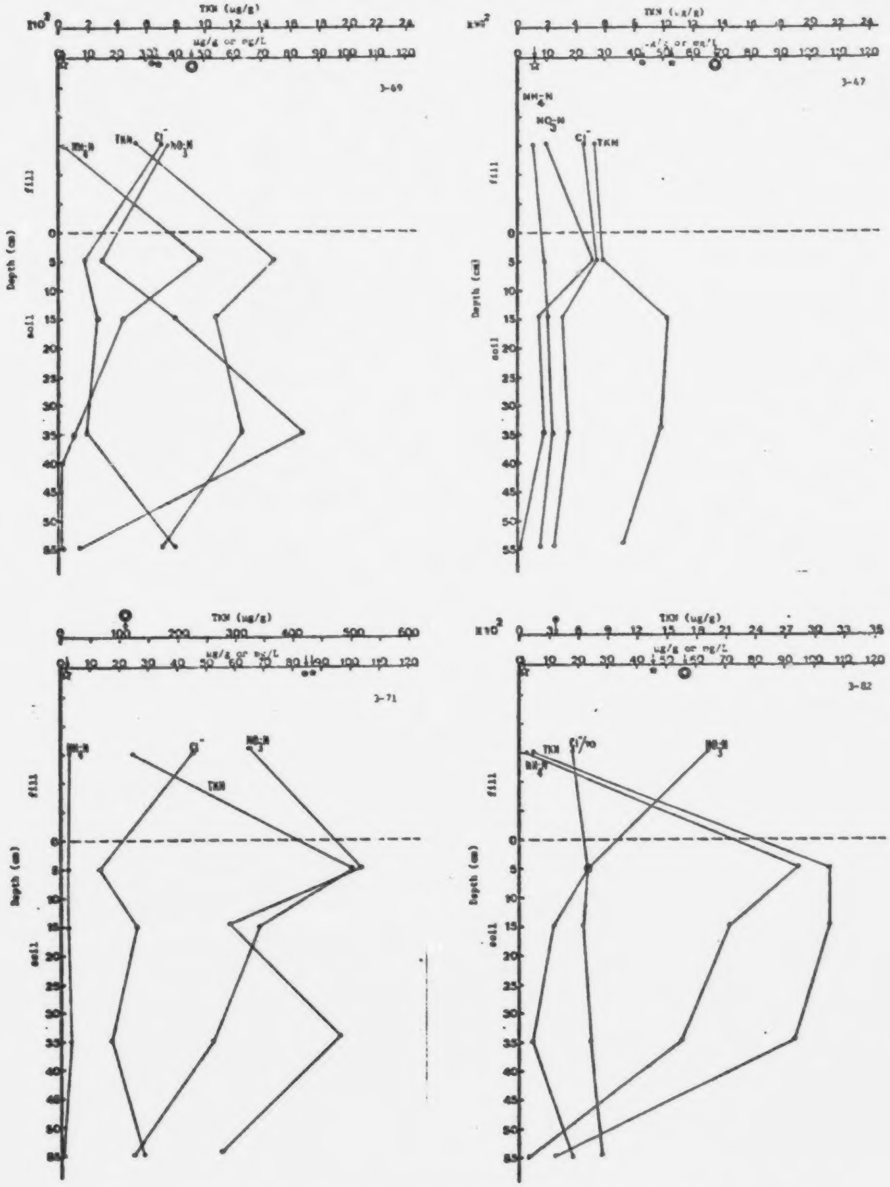

Fig. 18 . Average TKN, NH$_4$-N, NO$_3$-N and Cl$^-$ in the fill and natural soil under systems 3-69, 3-47, 3-71, 3-82.

⊙ = TKN, ★ = NH$_4$-N, ☆ = NO$_3$-N, ● = Cl$^-$ in the pumping chamber.

a horizon of a silty clay texture is encountered at the 35 cm depth.

In system 3-69, 64% of the NO_3-N formed in the sand fill is removed by immobilization and denitrification at the natural soil surface; however, at the 15 cm depth the system returns to the aerobic condition, allowing nitrification until the silty clay horizon is encountered at 35 cm. Denitrification then begins at the 35 cm depth due to the saturated anaerobic nature of the soil, denitrifying 93% of the nitrate present at the 55 cm depth (cf. Fig. 18).

At the soil surface in 3-69, NH_4-N is 30% enriched over the effluent level. The lower level of NH_4-N enrichment (compared to that in system 3-82) is due to the aerobic nature of the sand fill and the soil just below the surface, which allows more NH_4-N to be immobilized. The return to an aerobic condition is indicated by the rapid removal of NH_4-N below the 5 cm depth.

TKN values for system 3-69 show a maximum at the 5 cm depth and a reduction from 5 to 15 cm. Between 15 and 35 cm, the texture of the soil changes from a silt loam to a silty clay and permits concentration of organic N compounds at the surface of the silty clay horizon, presumably due to sorption on the clay and the anaerobic state of the soil.

Systems 3-47 and 3-71 experience extended resting periods between dosing events. Both systems exhibit excellent immobilization of NH_4-N with a higher level at the soil surface in system 3-47. This enrichment is due to the increased water content at the soil surface (15.5% for 3-47 vs. 7.2% for 3-71), which maintains an anaerobic state in the surface horizon of system 3-47 (cf. Fig. 18).

Systems 3-47 and 3-71 differ primarily in the NO_3-N concentrations at depth. System 3-71 shows 48% denitrification between 5 and 15 cm depth with a return to nitrification below 15 cm to 35 cm in depth. Starting at the 35 cm depth the system again begins denitrification and continues to the 55 cm depth denitrifying 42% of the nitrate existing at 35 cm. The higher water content of the soil and sand fill in system 3-47 reduces the nitrification level in the sand and promotes denitrification between 5 and 15 cm into the soil. Further N transformations below 15 cm have no net effect on the NO_3-N level with depth.

TKN values for each system show the same trend as the other parameters. System 3-71 deposits organic N compounds mainly at the soil surface. The aerobic state of the underlying soil is indicated by the constant reduction of organic N compounds with depth. System 3-47 deposits most organic N 15 cm into the soil where the water content and anaerobic conditions are greatest. The continued anaerobic nature is indicated by the slow reduction in organic N with depth.

Group 4 mounds (Table 16) all operate according to or close to recommended design specification (low or relatively low dosing rates, low uniformity coefficients) and manifest two beneficial results: immobilization of NH_4-N, preventing excessive enrichment at the soil surface, and denitrification as effluent passes into the soil profile (Fig. 19). This is due to the aerobic condition in the fill and the unsaturated flow through the sand.

All Group 4 mounds (Table 16) allow nitrification in the sand fill and subsequent denitrification of the effluent either at or just below

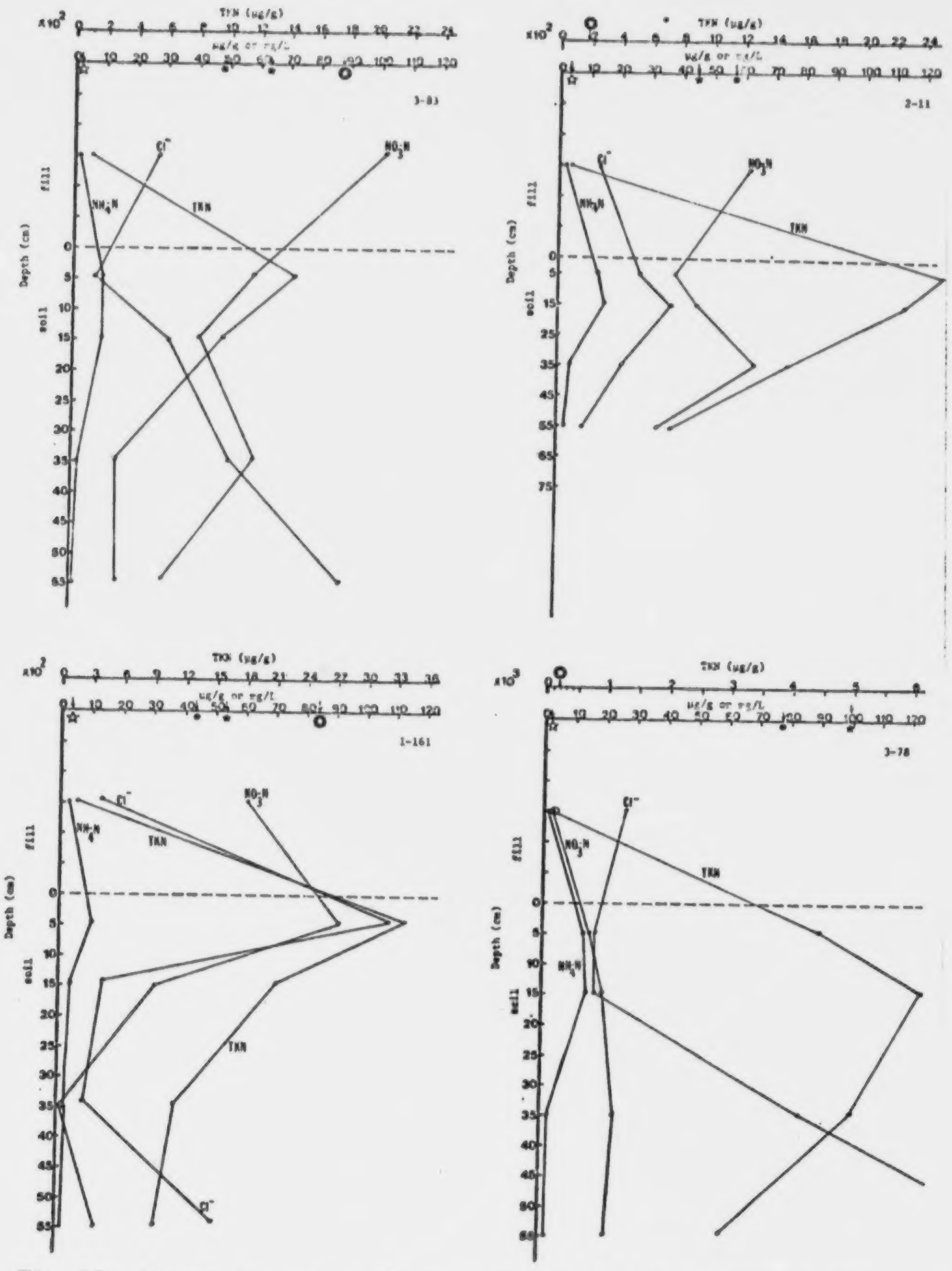

Fig. 19. Average TKN, NH_4-N, NO_3-N and Cl^- in the fill and natural soil under systems 3-83, 2-11, 1-161, 3-78.

✪ = TKN, ★ = NH_4-N, ☆ = NO_3-N, ● = Cl^- in the pumping chamber.

the soil surface (Fig. 19). System 3-83 is operating well within the guidelines suggested in the package design: NO_3-N concentrations in the soil solution within the sand fill are the highest by far for any system. At the soil surface, denitrification begins and continues 15 cm into the soil, removing 60% of the nitrate formed in the sand fill. Between 15 and 35 cm in depth, the soil becomes aerobic, and nitrification resumes. Between 35 and 45 cm a clay loam horizon is encountered, the soil again becomes anaerobic, and denitrification occurs removing 50% of the nitrate present at the 35 cm depth. TKN values indicate deposition of organic N compounds at the 5 cm depth in the soil with a continuous decrease with depth.

System 2-11, with 2 feet of sand, is operating in much the same manner as system 3-83, which has only 1 foot. Nitrification in the sand fill is followed by denitrification at the soil surface, reducing by 37% the NO_3-N formed in the fill. Mound 2-11 begins nitrification earlier in the soil (at 5 cm); nitrification continues to the 35 cm depth. Between 35 and 55 cm, denitrification again occurs removing 52% of the NO_3-N in the soil solution. The reappearance of denitrification is caused by a perched water table that fluctuated between 75 and 85 cm in depth.

System 1-161 has 1 foot of sand fill and a trench design. Nitrification begins in the sand fill and continues 5 cm into the soil. This situation is probably the result of a more even distribution of effluent by the trench design, which allows the soil to remain aerobic for at least 5 cm. Beyond 5 cm in depth, denitrification occurs and continues to 35 cm, removing 87% of the NO_3-N in the soil solution. This extensive denitrification is due to the fine texture of the soil (SiL to SiC) and the high water content which maintains the soil in an anaerobic condition.

System 3-78 in unique in two aspects: it serves a bar that operates full time in the summer but only on occasional weekends during the winter, and the soil underlying the mound is a fill comprised of various materials. Over a 2-year period, the system was dosed on the average once every 5.4 days, however reliable flow data for the summer are lacking due to counter malfunction, so the actual dose frequency while the bar is busiest is unknown. System 3-78 does not show nitrification within the sand fill and only minimal nitrification occurring 5 cm into the soil. Between 5 and 15 cm in depth little net change in NO_3-N occurs, however, below 15 cm in depth the NO_3^-/Cl^- ratios indicate denitrification, even though there is apparently a slight increase in total NO_3-N with depth.

Systems in Group 4 provide for immobilization and nitrification of fixed N forms in septic tank effluent within the sand fill and then denitrify NO_3-N at the natural soil surface. Thus they will lessen the problem of nitrate leaching into the groundwater by between 17 and 54% over the level predicted from work with laboratory columns, which exhibited a 33% loss of nitrate due to denitrification (Magdoff et al., 1974).

In summary, the mounds as presently operating exhibit a wide diversity in the manner and levels of N treatment, largely depending upon the dosing patterns being used. All systems reveal surprising capacities for immobilizing N in both the NH_4-N and organic N forms as well as for demineralizing by denitrification much of the NO_3-N formed in the mound. Mounds receiving frequent small doses of effluent are more efficient at immobilizing and nitrifying/denitrifying N inputs. Mounds operating according to recommended dosing specifications should not create problems due to NO_3-N levels in groundwater at present housing densities.

Phosphorus

All mounds

Figure 20 represents the changes with depth of the average total phosphorus (TP) and dissolved orthophosphorus (OP) values obtained from all of the systems monitored. There are two trends: OP reaches a maximum at 5 cm into the natural soil and decreases with depth, and TP reaches a maximum 15 cm into the soil and then decreases slowly with depth to the lowest level sampled.

Saturated soil moisture conditions increase the solubility of inorganic P compounds and retard microbial immobilization of P. In general, under present dosing practices, the top 5 cm of soils under mound systems is saturated so OP should reach a maximum or increase in this zone. In no system did the sand fill remove OP from the effluent before it reached the natural soil surface. The flow through the sand fill is either too fast to allow the sand to react with the OP and/or the sands are rapidly sealed by bacterial slimes, etc. that prevent them from reacting (cf. Sikora et al., 1976).

At 55 cm depth in the natural soil, OP values average 0.99 g/g, a value that suggests the soil solution OP concentrations may be far above

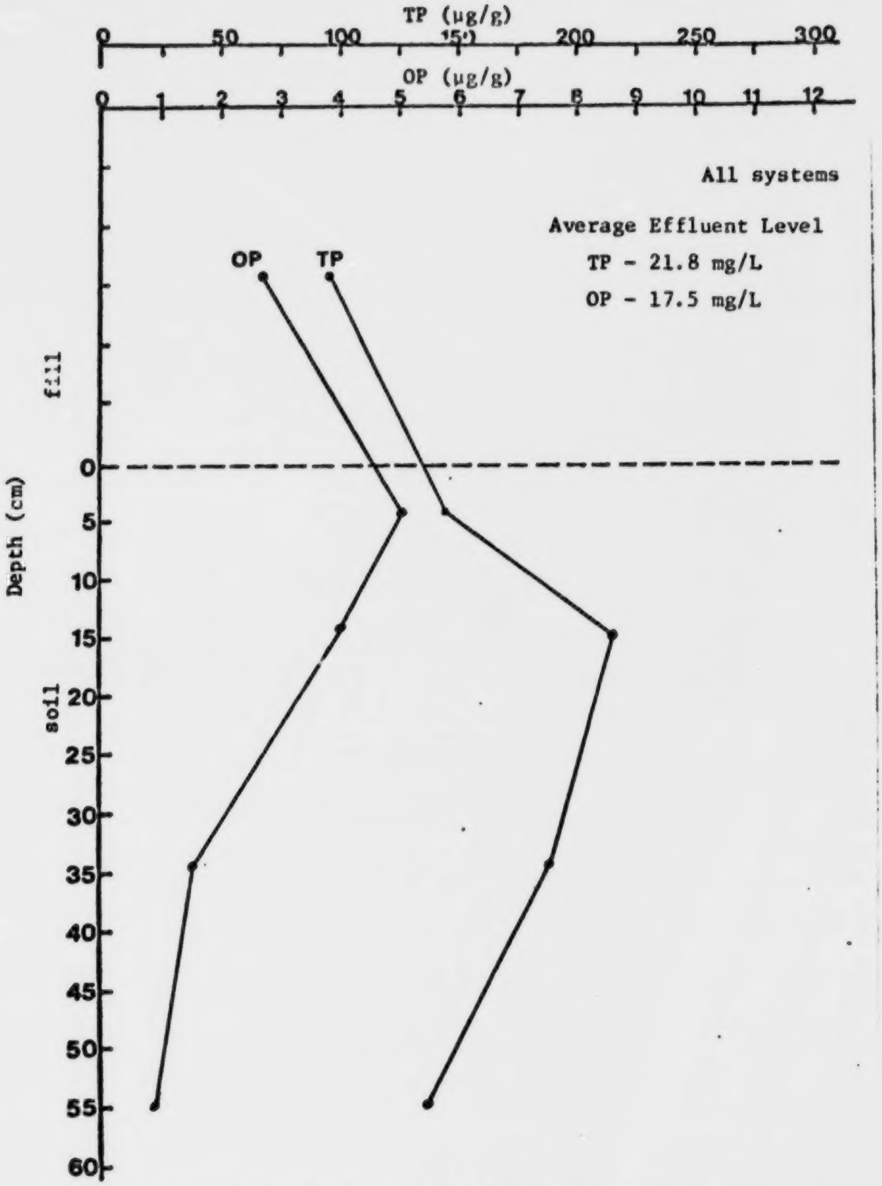

Fig. 20. Average TP and OP in the fill and natural soil under 33 mound systems.

the OP levels for groundwater (0-0.4 mg/L) found by Bouma et al. (1972). Because some mound systems are for soils above high groundwater or bedrock, high levels of OP might enter anaerobic groundwater, where further OP removal would be inefficient.

This high OP level is probably due to sealing of the soil by organic material which prevents OP removal by sorption onto soils, because TP does not reach a maximum at the soil surface, but rather 15 cm into the soil where the amount of organic material deposited by the effluent begins to decrease (cf. p. 132). The high dosing rates presently being used in most mounds are likely to flush OP rapidly through the fill and soil, regardless of whether soil particles are sealed with organics or not. Adequate times for sorption and precipitation reactions to occur are probably not being provided in many cases.

Individual package differences

Phosphorus behavior for the various package designs (Fig. 21) indicates that Package 1 and 2 designs remove more OP than Package 3 designs, and that Package 2 designs show higher TP levels in the sand fill than Package 3 or Package 1 designs. Removal of OP by the Package 1 and 2 designs is probably generally valid, even though Package 2 design data include values from two systems (2-11, 2-105) which gave excessively high values for all parameters monitored, tending to skew the data to higher values. However, samples from Package 2 systems at the 35 and 55 cm depths were obtained only from systems 2-11, 2-105, and 2-243 because rocks and cobbles prevented sampling the other Package 2 systems at these depths.

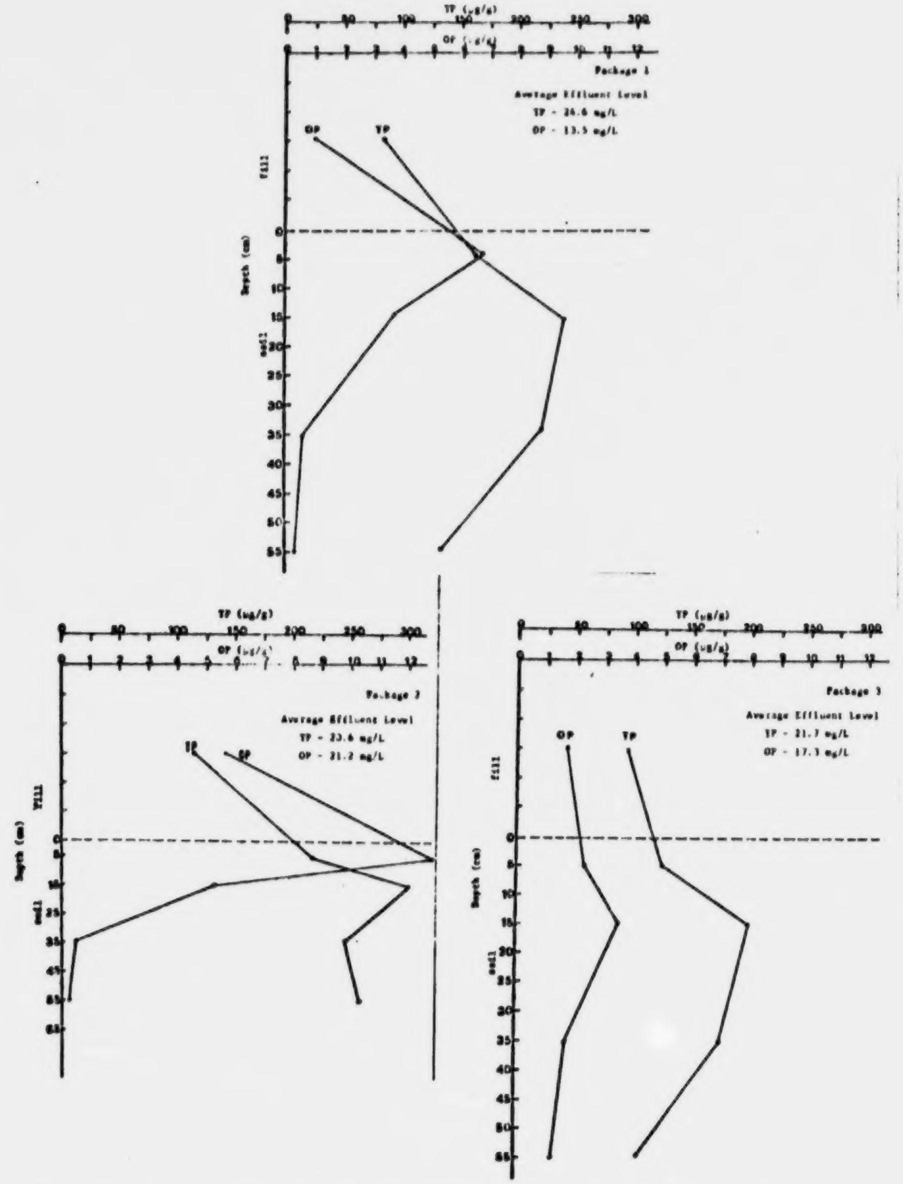

Fig. 21. Average TP and OP in the fill and natural soil under package 1, 2 and 3 mound systems.

The good removal of OP in Package 1 systems is due to the trench design of the dosing system and to the fact that they overlie slowly permeable soils, which have a high clay content and which therefore sorb P better than permeable sandy soils. The removal of OP in Package 2 systems is enhanced by the 2 feet of sand fill incorporated into the design, since this increases the treatment depth for P.

Package 3 mound systems show the highest OP levels at the 55 cm depth (1.19 vs. 0.40 vs. 0.34 g/g for Packages 3, 1, and 2 designs). This is important since Package 3 is designed for soil with high groundwater (2 ft minimum). Thus this level of OP could exist only 5 cm above a water table which would limit the further removal of P from the effluent.

Phosphorus movement in individual systems

The discussion of P movement for selected individual systems follows the format described for bacterial attenuation (see p. 88 and Table 16).

The major differences between the two Group 1 systems considered (3-34 and 3-64 in Table 16) is the dose frequency: system 3-34 is dosed on the average once every week and system 3-64 once every day. System 3-34 sustains quasi-saturated flow in the sand fill immediately after dosing, but there is a six-day resting period between doses; system 3-64 maintains saturated or almost saturated conditions constantly. Figure 22 shows a dramatic difference in their respective TP and OP values. In system 3-34, resting between doses allows for the return of aerobic conditions and fosters excellent removal of OP (reduced to 0.05 g/g by 55 cm into the soil). System 3-64 exhibits much higher OP value at all

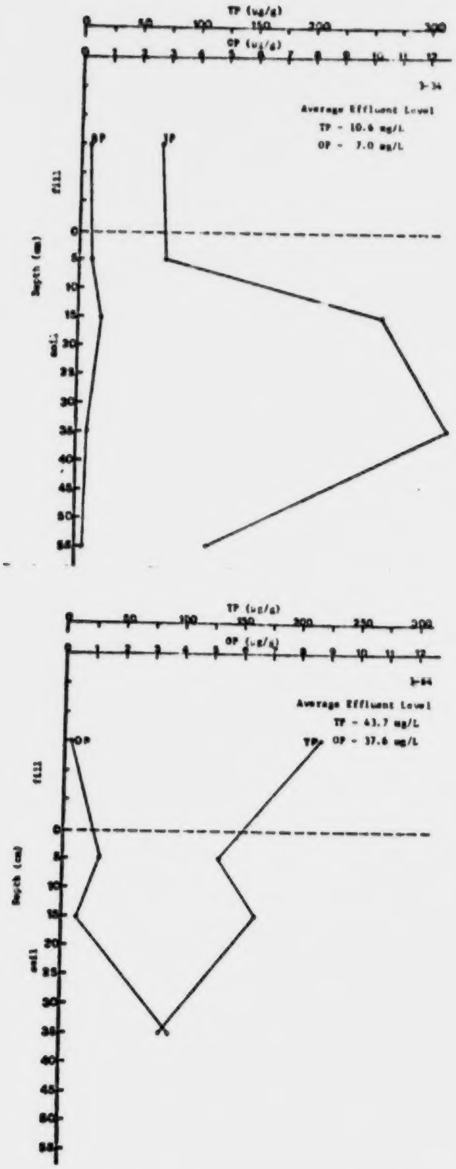

Fig. 22. Average TP and OP in the fill and natural soil under systems 3-34 and 3-64.

depths. A surprisingly high OP value occurred at the 45 cm depth (3.5 µg/g) where the soil horizon shows a silt loam to silty clay texture change. Samples below 45 cm could not be obtained because of a saturated zone at 45 to 50 cm.

Of the Group 2 systems (Table 16), mound 1-160, with its trench design in the distribution network, shows better removal of OP before effluent enters the natural soil; because the system returns to an aerobic state at 55 cm depth, the OP level is below the detection limit (0.02 µg/g) at this depth (Fig. 23). System 3-61 differs markedly, having essentially saturated conditions throughout the soil profile. Three times during 2 years a saturated zone was observed at 50 cm depth; as a result the OP level at 55 cm is the highest value obtained from any mound at this depth. This reaffirms the tendency of saturated conditions to increase OP levels and the potential for P movement into groundwater.

Systems in Group 3 (Table 16) all exhibit an aerobic state at some depth in the soil profile; at this position there is a rapid decrease in OP levels (Fig. 24). The average OP level at the 55 cm depth, the P that can potentially enter groundwater,is 0.44 µg/g. Excluding system 3-71, this average would be 0.24 µg/g. System 3-71 is located over a well aerated sandy loam with a percolation rate of 1 min/in; water transport through this profile is rapid, reducing the time availble for P removal by immobilization in bacterial cell mass or soil sorption and precipitation reactions.

The rapid percolation rate of the soil under system 3-71 is also reflected in the TP figures, which reach a maximum at a 5 cm depth in

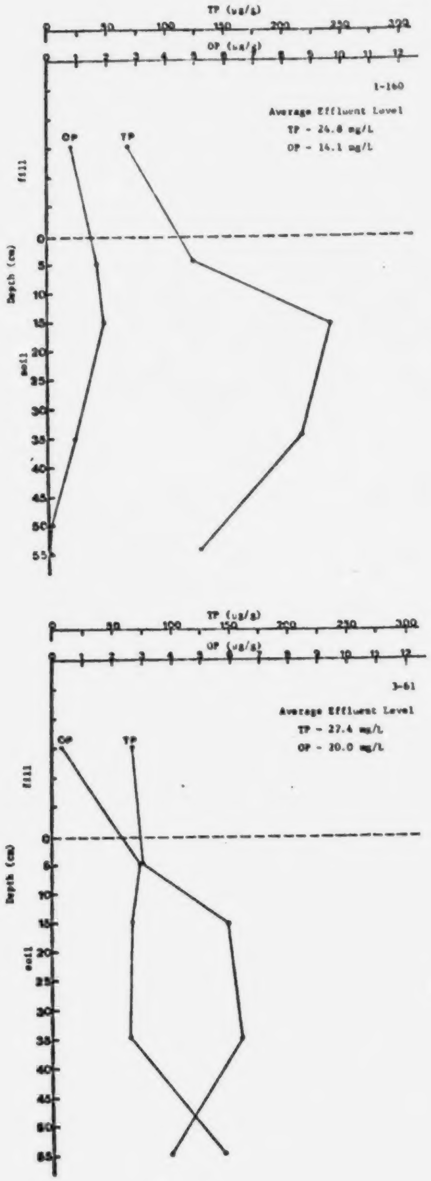

Fig. 23 . Average TP and OP in the fill and natural soil under systems 1-160 and 3-61.

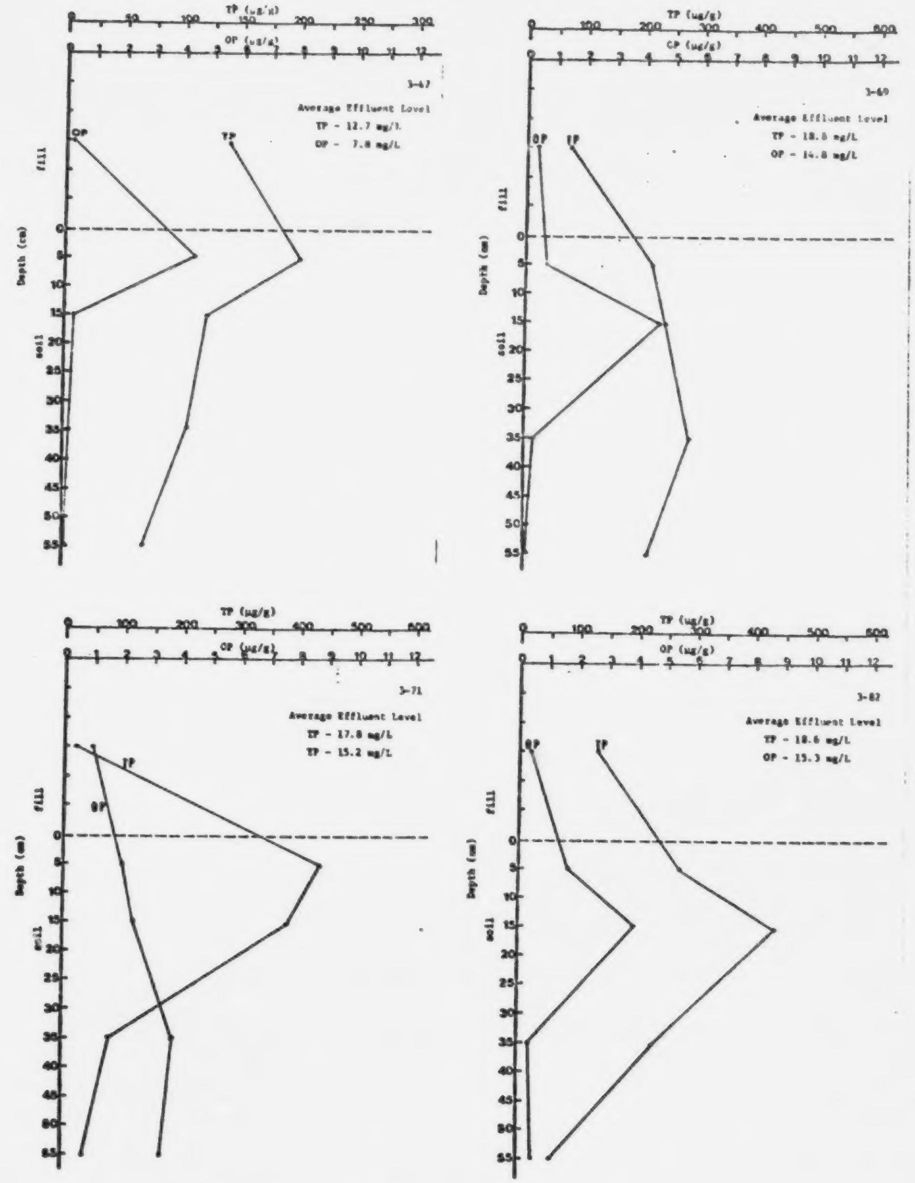

Fig. 24. Average TP and OP in the fill and natural soil under systems 3-47, 3-69, 3-71 and 3-82.

the soil. System 3-69 shows a maximum TP value at 35 cm depth into the soil even though the soil texture is less coarse than that for systems 3-71. System 3-82 is located over Plainfield loamy fine sand with a percolation rate of 5 min/in. This system doses much more frequently than systems 3-69 and 3-71. The rapid dose frequency of system 3-82 apparently prevents OP movement deeper into the soil profile.

The P results for systems in Group 4 (Table 16) are shown in Fig. 25. One system in Group 4 contains some suspect data. For 6 of 8 sampling periods, system 3-83 showed typically low P values in the soil; during the last two samplings periods the OP values increased dramatically. For example, at the 15 cm depth the average for 6 sampling periods was 0.45 g/g, but for March and June 1978 the average value rose to 27.38 ug/g. The only unusual development was a change in home ownership just before March 1978, but no significant change in the septic tank effluent P levels accompanied the increase in soil solution. Even with the extraordinary high P levels in the overlying soil, the level in soil at the 55 cm depth was reduced to 1.8 ug/g, a 93% reduction.

System 2-11 is typical of all but one of the Package 2 systems studied. The TP values increased steadily with depth reaching a maximum at the 55 cm depth into the soil. Most Package 2 systems had the highest concentrations of P in the septic tank effluent, in addition the soils below the Package 2 systems are loams (Longrie or Summerville loam) of high water content. Thus it appears that P is percolating through the anaerobic surface soil horizons and participating in precipitation reactions or microbial immobilization only after the

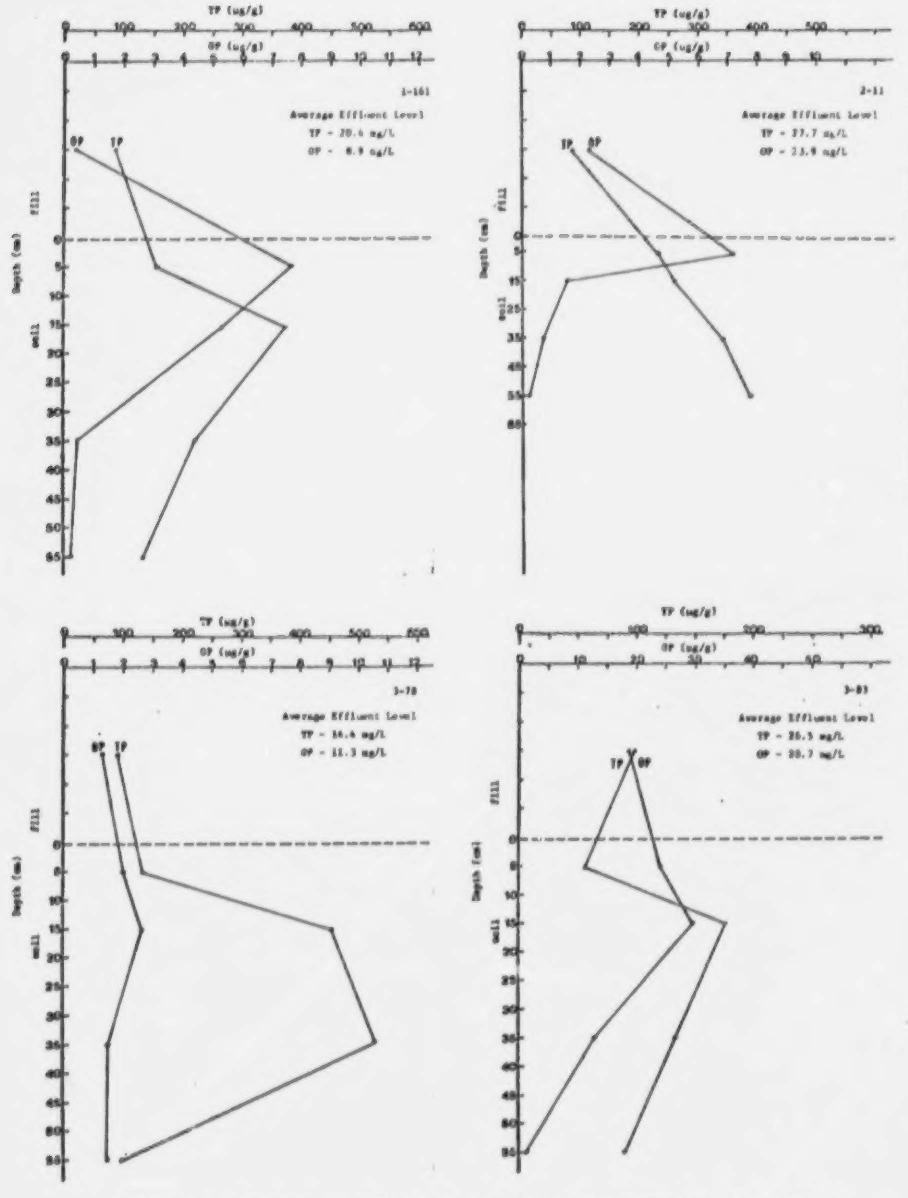

Fig. 25. Average TP and OP in the fill and natural soil under systems 1-161, 2-11, 3-78 and 3-83.

effluent enters the aerobic subsoil. As with all the systems studied, the OP values reach a maximum at or near the soil surface where the soil was at or near saturation.

Typically, the soil OP is highest where saturated conditions prevail in soils. Systems 1-161 and 3-78 both exhibited the highest OP values where the highest water content occurred in the soil profile (5 cm for 1-161 and 15 cm for 3-78; cf. Table 10).

The TP values for 3-78 reach a maximum at the 35 cm depth in the soil, but because much of the soil beneath this system is fill material, no conclusion can be made, except that here the TP maximum occurs after the saturated surface layers of the soil. The explanation may lie in highly variable native TP levels in the fill material underlying this mound.

The soils below all the mound systems exhibit a capacity to immobilize much more P than would be indicated by a Langmuir isotherm; P immobilization is occurring even after soil exchange sites have become saturated. Such P removal is due either to inorganic precipitation reactions or immobilization into organic material (Sikora and Corey, 1976). High TP values observed in systems 3-78 (507.3 µg/g at 35 cm, Keowns silt loam), 3-82 (412.6 µg/g at 15 cm, Plainfield loamy fine sand), and 1-161 (360.2 µg/g at 15 cm, Kewaunee silt loam) prove that the soil immobilizes much higher levels of P than a Langmuir absorption treatment would indicate, supporting conclusions of Magdoff and Keeney (1976) who found immobilization of P levels of 307 µg/g in columns representing mound disposal systems.

For purposes of comparison, OP and TP values in samples of surface soils collected in the vicinity of several of the mounds studied are shown in Table 18. These values indicate that P enrichment is occurring with time in the soils underneath the mounds to levels above the ambient levels in native soils.

Thus, the mounds are removing substantial amounts of P from the percolating wastewaters. The sites and extents of P removal depend mainly on the type of soil under each mound and the dosing regime that is being used. Dosing conditions approximating those prescribed in the mound construction packages foster good P removal by both biological and chemical reactions as effluent percolates slowly under unsaturated flow conditions through the mound.

Seasonal variations in mound performance

No distinct seasonal trends in bacterial indicators occurred; most systems maintained a consistent number of the organisms tested throughout the year. Since several systems were underused during the winter, this could indicate preservation of indicator organisms in soil during the colder months. Gerba et al. (1975) have shown that, after removal from sewage water, bacteria may survive in soil from a few hours to several months, depending on environmental conditions.

In general, TOC and TKN show rapid increases in the sand fill and in the first 15 cm of soil during the winter months. Enrichment of these parameters deeper in the soil occurred during the summer months. Phosphorus analyses indicated that TP values increased steadily throughout the year.

Figure 26 shows the average values of TOC, TKN and TP obtained from all systems from October 1977 to September 1978. In the sand fill TOC

Table 18. TP and OP in g/g of dry soil, for soil samples collected at a 5 cm depth from areas surrounding several mounds.

System	TP	OP
1-40	87.8	0.25
1-99	111.5	0.20
1-160	--	0.0
1-161	137.0	0.0
2-11	161.5	--
2-12	--	0.35
2-76	132	0.0
2-105	111.2	1.0
2-109	103.0	0.0
2-243	121.0	0.0
3-24	127.0	0.52
3-47	19.1	0.0
3-54	71.1	0.0
3-61	--	0.0
3-64	51.1	0.20
3-69	159.9	0.25
3-71	77.1	0.0
3-72	77.2	0.0
3-73	56 1	--
3-74	47.2	0.41
3-78	101.5	0.0
3-83	27.5	0.51
3-110a	47.5	0.20
3-110b	52.2	0.75
3-112	31.1	0.70
3-123	74.4	0.0
3-234	33.4	0.0
3-273	23.1	3.05
n	25	26
Arithmetic mean	81.6	0.32
Geometric mean	69.4	0.0007
Std. Deviation	43.1	0.62
Standard Error	8.6	0.12
95% Confidence	98.5	0.12
Interval	64.7	0.08

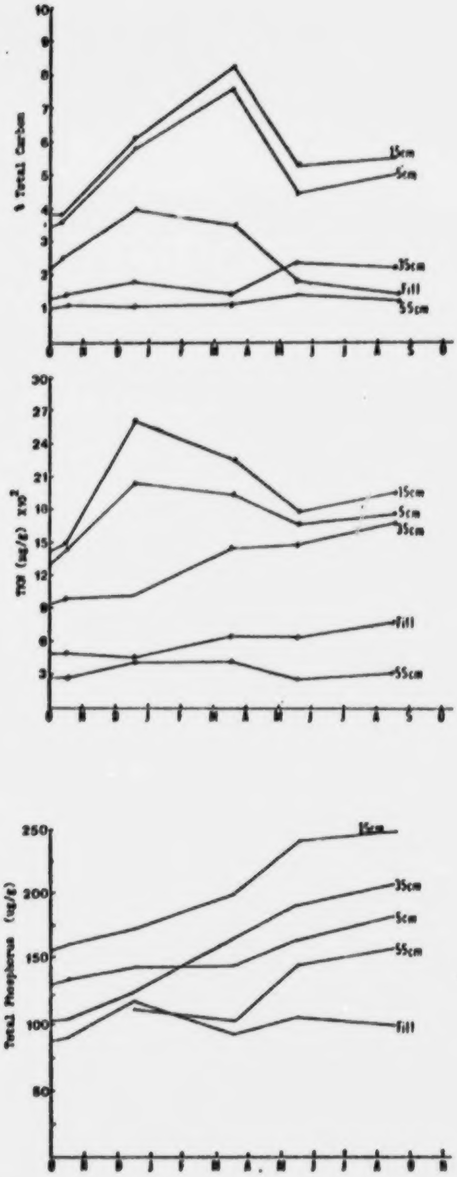

Fig. 26. Average TOC, TKN and TP in the fill and natural soil under all systems sampled from Oct. 1977 to Aug. 1978.

and TKN increase during the winter, but return to approximately the same levels during warmer weather. The values in the soil also fall to a lower level in the summer, but show a net increase over the two-year sampling period. Similar observations have been made on small model mounds in Vermont (Simons and Magdoff, 1979a). Soil TP levels increased steadily throughout the sampling period; in the sand fill, TP returned to the levels of the previous summer. After the winter, upon return to warmer weather, TKN decreased sooner than TOC. This is normal as bacteria incorporate N quickly when there is abundant organic carbon available (Mitchell, 1972; Alexander, 1977).

The most critical period for a mound is during the winter months. Because of the exposed nature of the mounds, their temperatures drop, reducing microbial metabolism and allowing for deposition rather than degradation of wastes. If, as now recommended (cf. p. 163), the dose volumes are lowered and coupled with a decreased time between dosing events, the more frequent flushing of the mound with fresh warm effluent would serve to raise the internal temperature and prevent deposition of material that may lead to clogging and failure. Clogging and failure of soil columns is accelerated by low temperatures (Simon and Magdoff, 1979b). Wintertime accumulations of undigested wastes may have compounded the problems of hydraulic overloading and inappropriate fill texture that led to failure of the three mounds that allowed surfacing of effluent during the winter (see p. 61).

Clogging mat formation

After completion of the two-year monitoring program, an attempt was made during the spring of 1979 to determine whether a clogging mat was forming in 4 systems (3-47, 3-72, 3-78, and 3-82) by sampling at the same depths as before and at the gravel/sand fill interface. This was the only time samples were taken from this interface. All samples were analyzed for TOC and TKN.

Table 19 shows the carbon:nitrogen (C:N) ratio for the values obtained, with the dosing rates and uniformity coefficient values for the respective sand fills. Figure 27 shows the TOC and TKN values obtained during this sampling. The C:N ratio is one means of expressing the biodegradability of organic materials: a low C:N ratio (<20) indicates that organic material is readily biodegradable, a high C:N ratio (>20) indicates it is resistant to breakdown by soil microorganisms (Foth and Turk, 1972). The C:N ratio for septic tank effluent and soil organic matter is approximately 10 (Sikora and Corey, 1976).

Since most of the N in septic tank effluent is in the NH_4-N form and is not associated or combined with the high carbon material (vegetable residues, toilet paper, etc.), the increase in the C:N ratio at the gravel/sand interface indicates deposition of high carbon wastes while N in soluble ionic forms is being transported with the percolating effluent into and below the fill.

System 3-47 has a high dosing rate and a low uniformity coefficient for the sand fill (Table 19). The C:N ratios indicate movement of much of the organic matter from the septic tank effluent is occurring: the maximum values for both TOC and TN are at 15 cm depth into the natural soil (Alcona sandy loam). The C:N ratio also reaches a maximum at the 15 cm depth. This suggests that systems that are being overloaded with

Table 19. Total carbon:total nitrogen ratios for soils in and below four mound systems.

Horizon	Mound 3-47	 3-72	 3-78	 3-82
Gravel/sand interface	35.2	59.12	241.1	110.36
Sand fill	16.04	19	194.3	103.6
5 cm in soil	58.41	40	18.2	21.1
15 cm in soil	75.8	34	16.4	23.4
35 cm in soil	61.4	12	16.5	9.96
55 cm in soil	22.6	9	9.7	17.46
Dosing rate (gal/ft^2/dose)	0.95	0.76	0.65	0.86
Uniformity coefficient	3.7	6.4	2.5	3.0
% Fines (silt and clay)	1.8%	3.8%	2.4%	3.0%

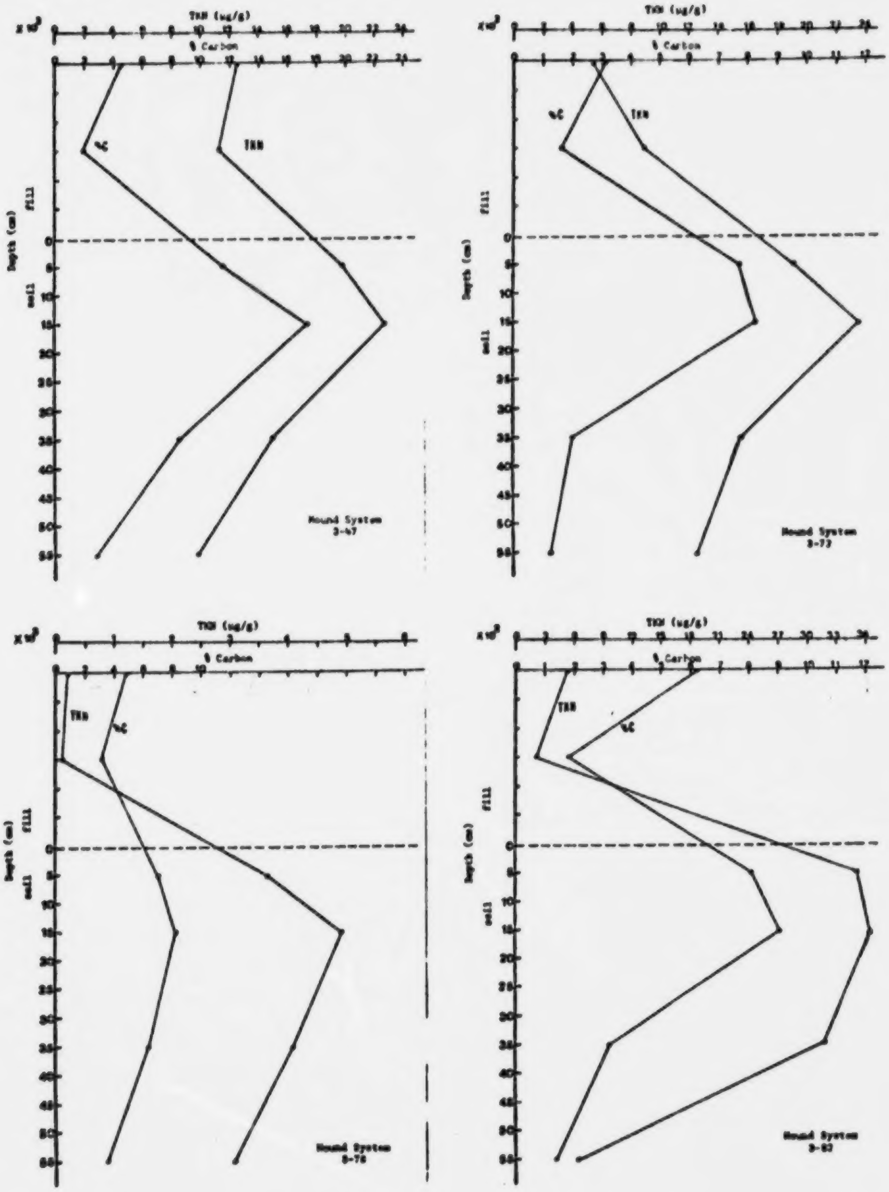

Fig. 27. TKN and TOC at the gravel-sand interface and in the fill and natural soil under systems 3-47, 3-72, 3-78 and 3-82.

each dose may form an organic clogging mat at the sand fill/soil interface or just into the soil.

System 3-72 also has a high dosing rate but the uniformity coefficient of its sand fill is higher (Table 19). A much different picture results for the C:N ratio profile. The maximum occurs at the gravel/sand interface with a decreasing C:N ratio in the soil. The C:N ratio of the soil organic matter below this system is more typical of that of soil microbial tissue, which is approximately 10 (Alexander, 1977). The more bioresistant material is being deposited at the top of the sand fill indicating that the organic clogging mat is being formed at this level, and not in the soil as with system 3-47.

System 3-78, discussed previously, is at a bar that is open only during the summer and on winter weekends. As with system 3-72, the TKN and TN are highest in the natural soil, but the maximum C:N ratio exists on top of the sand fill and in the sands. This again indicates that any organic crust forming is at the gravel/sand interface. System 3-82 is similar; the maximum C:N ratio is at the gravel/sand interface indicating that a clogging mat is probably forming atop the sand fill.

The development of a mature crust could change dramatically the mound performance described so far. A mature crust on top of the sand would pond effluent in the gravel bed and produce unsaturated flow through the sand fill eliminating the surge effect of the dosing system that carries bacteria and nutrients to lower levels in the seepage area in systems receiving large doses. In addition, by slowing the infiltration rate into the sand fill, the saturated zone at the soil surface may be eliminated, reducing the amount of denitrification that is occurring in many systems at or near the soil surface. Overall, crust formation at the top of the sand fill would cause the mound systems to function more like a conventional soil absorption field (cf. p. 146).

Comparison to other systems

Bacterial attenuation by soil

Bacteriological data collected during mound monitoring were obtained from soil samples; thus, comparisons with data from other studies which use leachate samples must be made with caution. Leachate samples tend to give lower results for all parameters, both microbial and chemical, than the soil samples obtained in field monitoring of mounds. Soil samples contain the total amounts present of any material or organism assayed while leachate samples show only those species that are in aqueous suspension or solution and not those sorbed onto or associated with the soil particles. In addition, most *in vitro* studies with soil columns (lysimeters) use septic tank effluent that has been removed from the septic tank and stored, usually in a refrigerated condition. Indicator organism counts in stored septic tank effluent are often 50-60% lower than those in the fresh effluent analysed by the monitoring project (Iskandar, 1979); these organisms have a finite survival time outside the human body and many die off even before being applied to laboratory columns.

In studies using 50 cm columns of sand and silt loam soils, Ziebell et al. (1975) found results similar to those of field monitoring of mounds. The mound systems performed well in the removal of both TC and FC before the natural soil surface, achieving on the average 99.5% TC reduction (dry weight basis-99.99%) and 92.8% FC reduction (dry weight basis-99.9%). Ziebell et al. (1975) found 94.1% TC and 99.9% FC reduction in stored septic tank effluent after filtration through 50 cm of sand in columns dosed at 2 to 10 cm/day (0.48 to 2.4 gal/ft^2/day). Similar results were obtained on columns of silt loam with increased levels of TC and FC; however, the silt loam columns, like the mound system, allow FS and *Ps. a.* to move with much less reduction. In fact, like samples from deeper horizons in the mound systems, the effluent from the columns contained more *Ps. a.* than were in the initial septic tank effluent applied. Ziebell et al. (1975) concluded that *Ps. a.* multiplied within their columns, as conditions in the silt loam were satisfactory for growth. This is reasonable since *Ps. a.* is a voracious heterotroph capable of growth on a large variety of carbon sources and able to use NO_3- as a terminal electron acceptor in the absence of oxygen (Hoadley, 1976).

The literature contains no information on movement of *Ps. a.* through soil. It therefore seems that the excessive level of *Ps. a.* in the natural soil under some mounds is not due to lack of efficiency in the mound design to remove *Ps. a.*, but rather to the multiplication of these bacteria after they enter the soil.

Column studies have also demonstrated the ability of FS to move through columns of sand and silt loam soils (Ziebell et al., 1975). The mound study indicates minimal reduction of FS in soil samples taken at the natural soil surface underneath mounds. However, if the bacterial numbers are calculated on a dry weight of soil as in most other literature, FS are 71% removed before they reach the natural soil surface. Ziebell et al. (1975) observed FS movement in a sand column dosed with 10 cm/day of septic tank effluent but not in a column loaded at 5 cm/day (2.4 and 1.2 gal/day/ft^2, respectively). Similar results were reported for silt loam columns dosed at 1 cm/day (0.24 gal/day/ft^2); columm leachate contained up to 1 x 10^4 FS/100 ml. By lowering the loading rate in these columns from 1 cm/day to 1 mm/day (0.024 gal/day/ft^2), the movement of all fecal indicator bacteria was essentially stopped.

This indicates that the FS movement observed in mound systems is to be expected at the loading or dosing rates presently being used in many mound systems even if the fill is of the proper texture and uniformity. Indeed, FS averaged 1.8 x 10^3 MPN/100 g of dry soil or 8.5 x 10^3 MPN/100 ml of soil solution at the natural soil surface below mound systems. However counts in this order of magnitude are often encountered in surface soils not receiving septic tank effluent (cf. Table 15).

Lower dosing rates move bacteria more easily through silt loam than through sand in columns. Similarly, lower dosing rates move bacteria more easily through finer fill with high uniformity coefficients in mounds.

The fill material and its dosing system are analogous to a rapid sand filter. Filters with an effective sand size of 0.2 mm and a uniformity coefficient of 3-4 reduced TC, FC, and FS in septic tank effluent one hundredfold (2 logs) when loaded for 30 days at 20 cm/day (5 gal/day/ft^2) but afforded effluent of unacceptable bacterial quality (approx. 10^3 TC, 10^3 FC, 2.5-8.0 x 10^2 FS per 100 ml) even though the filters clogged and failed rapidly (Sauer et al., 1976). However here

Ps. a. counts were reduced by 2-3 log factors, demonstrating the efficiency of a clogged soil to remove these mobile organisms. These conditions are not easily comparable with those in a mound. Converse et al. (1974) studied sand filters 2 feet thick with a sand uniformity coefficient of 3.0 and determined TC, FC and FS counts in the filter effluent. A filter loaded at 7.5 cm/day (1.8 gal/day/ft^2) removed 98% TC, 99% FC, and 96% TC. A filter loaded at 15 cm/day (3.6 gal/day/ft^2) removed 90% TC, 95% FC, and 69% FS. These results were based on 3 leachate samples collected once every 3 months. The filters were dosed only once a day but with stored septic tank effluent that had only half the number of indicator organisms in average effluent from mound systems.

Because they remain unclogged, on the average, mound systems seem less efficient than rapid sand filters with sand of 3.0 uniformity coefficient, even though as constructed, mound systems had higher uniformity coefficients in the sand fills (average 5.78). However, bacterial counts in mounds were performed on soil samples rather than leachates. Moreover, in the discussion on bacterial attenuation in individual mound systems (pp. 94), mound systems in Group 4 (average uniformity coefficient = 2.48) reduced TC and FC by over 99%. Thus, mounds constructed and used according to design specifications reduce coliforms better than rapid sand filters.

In studies of experimental mound systems (Bouma et al., 1975), samples of mound seepage collected above slowly permeable natural soils in 3 systems loaded at rates of 0.35-0.72 gal/day/ft^2 were examined. This liquid contained 240-1700 TC, 50-580 FC and 60-150 FS per 100 ml, even though lateral movement through the fill had occurred to the point at the toe of the mound where the samples were taken. These values represent attentuation ratios of 3471-7917 for TC, 1017-12000 for FC but only 3.0-400 for FS when compared to the average septic tank effluent TC, FC and FS levels. Dosing rates or dosing frequencies were not measured for these systems, but the fact that water ponded above the natural soil suggests that all were being overdosed. A fourth mound, constructed using techniques now recommended for Package 1 designs (Converse et al., 1976a; Converse, 1978) did not pond above the natural soil, although loaded at a rate of 0.92 gal/day/ft^2. Consequently no bacteria counts were obtained, since methods for sampling soil from beneath mound gravel beds had not been evolved. Similar levels of bacterial attenuation had earlier been observed in experimental mounds built over creviced bedrock (Bouma et al., 1972). In general, the mounds built by commercial installers that have now been examined apparently remove bacteria as efficiently (Tables 10-12) as the experimental systems installed and monitored by the scientists who developed the design now used.

In conventional septic systems, effluent trickles out of the septic tank and flows by gravity into a gravel trench or soil absorption field in the same random pattern as fresh wastewater enters the tank. As a result of this uneven flow, an organic crust rapidly forms around the absorption area and tends to pond effluent in the trench (Harkin et al., 1976). In time, the crust slows infiltration into the natural soil and allows only unsaturated flow beyond the crust. The crust also serves as an excellent filter medium for the removal of bacteria. Bacterial attenuation in a conventional septic system thus depends upon the flow regime (saturated or unsaturated flow), the texture of the soil, and the hydraulic resistance of any biological crust present (Bouma et al., 1972; Bouma, 1975).

No mound system has yet developed a mature crust at the gravel/sand fill interface or at the sand fill/natural soil interface and thus a potential for saturated flow within and beneath the mound systems persists. The lack of a crust reduces the filtering ability of the natural soil or the sand fill. In their present condition, mound systems are therefore comparable to newly installed conventional septic systems.

Studies on conventional septic systems (Bouma et al., 1972; Walker et al., 1973a,b; Tyler et al., 1977; Brown et al., 1977, 1978) have produced results that are highly variable; bacteria counts are not always reported. Counts reported have usually been performed on soil or leachate samples but the assays used have not always been sufficiently sensitive to afford meaningful results.

Brown et al. (1977, 1978) studied the movement with time of fecal coliforms through 3 soils, a clay, a clay loam and a sandy loam and examined both soil and leachate samples, reporting MPNs of FC per gram of dry soil or ml of leachate. In the clay loam and sandy loam, FC travel into the soil: up to 4.3×10^4 FC/100 g of soil were found 80 cm from the absorption bed in the clay loam soil, up to 4.0×10^5 FC/100 g of soil 25 cm from the absorption bed in the sandy loam soil. After 2 years, bacterial attenuation in the clay loam improved with significant FC movement occurring only laterally from the absorption bed (1.7×10^4

FC/100 g 20 cm from the absorption bed). Presumably a crust had formed and reduced the downward movement of FC. In the sandy loam FC travel persisted: after 2 years 1.0×10^5 FC/100 g of dry soil occurred 30 cm below the absorption bed.

On the average mound systems reduced FC to 2.3×10^3 MPN/100 g of dry soil (6.8×10^3 FC/100 ml of soil solution) by the natural soil surface. These values are close to the background level found in many surface soils (Table 15) and represent better FC attenuation than that found by Brown et al. (1977,1978) for conventional systems in clay loam and sandy loam soils.

Bouma et al. (1972) performed a field evaluation of 19 conventional and 6 experimental mound systems in some major soils in Wisconsin, but used plate counts for bacterial assays; counts of less than 1 TC, FC or FS per ml of leachate or 10-100 TC, FC, or FS per g of moist soil (10-20% moisture content could not be determined by this method. Bouma et al. (1972) concluded that there was little bacterial travel from crusted systems which permitted only unsaturated flow in the soil. However, 2 of the 19 conventional systems had incomplete crusts with saturated flow away from the absorption bed; both systems showed extensive contamination of groundwater by TC, FC, and FS. Soil bacterial levels 30 cm away from clogged conventional systems were found to be <600-23000 TC, <200-17000 FC and <200 FS/100 g of moist soil (Ziebell et al., 1975).

Because differences in soil and site conditions, sampling procedures and bacteriological testing and data reporting complicate comparisons of earlier studies with the results of the present mound monitoring, two ponded (i.e. crusted) conventional systems in sandy soils were therefore sampled with the techniques used to obtain samples from mounds. The samples were then analyzed for TC, FC and FS by the same methods as were used for fill and soil samples taken from mound. The results are shown in Table 20. Both systems were located in Plainfield sand with the same texture and uniformity as the material recommended for mound fill.

On the average mound systems reduce all fecal indicator bacteria numbers below 10^3-10^4 per 100 ml of soil solution in the soil under the seepage area, indicating that even though many mounds are being overdosed, on the average they reduce bacterial population as well as or better than some clogged conventional systems (cf. Table 20).

The only possible bacteriological concern with the mound systems as now operating is the movement of FS below some systems, due to the excessive dose volumes that promote saturated flow through the fill and into the natural soil in these systems. Some pathogenic bacteria in wastewater might move in the same way as FS. However, conventional systems over sandy soil obviously also allow FS to travel into the soil.

After two or more years of operation some heavily loaded mound systems appear to be slowly forming a crust at the gravel/sand fill interface. Should crust buildup continue, bacterial attenuation would match or exceed that in conventional systems as the flow regime dictated by a crust in the sand fill coupled with dosing would prevent most bacteria from entering the natural soil. In systems without a crust, lowering of the dose volumes to recommended levels would markedly improve bacterial attenuation.

Table 20. Bacterial counts (TC, FC, FS) obtained below two conventional wastewater disposal systems in coarse-textured soils (Plainfield sand)

Depth below absorption field	MPN/100 ml of soil solution		
	TC	FC	FS
	Pine Vista Mobile Home Park Columbia County, Wisconsin n=2		
5 cm	780	120	6.0×10^3
15 cm	1.6×10^4	120	6.0×10^3
35 cm	1.5×10^3	240	3.3×10^4
55 cm	9.6×10^4	9.6×10^4	6.0×10^3
	Lake Camelot Adams County, Wisconsin n=1		
5 cm	2.4×10^4	5.4×10^3	4.1×10^4
15 cm	1.6×10^4	1.1×10^3	7.5×10^3
35 cm	1.6×10^4	9.2×10^3	3.0×10^3
55 cm	3.5×10^3	2.4×10^3	4.0×10^3

Nitrate movement

Nitrate is an important concern with all soil absorption systems, as NO_3-N leaching to groundwater may cause public health and environmental problems (Comly, 1945; Gruener and Shuval, 1970; Winton et al., 1971; Shuval and Gruener, 1972; Delfino, 1977). It has been assumed in the past that eventually all the N entering a conventional waste disposal system will be converted to NO_3-N (Walker et al., 1973b). However, Magdoff et al. (1974) concluded from laboratory studies that approximately one-third of the NO_3-N formed in mounds is denitrified, thus reducing the input to ground or surface waters by one-third from the total N input of the system. In practice, the amount of denitrification in mounds seems to be higher.

On the basis of data summarized in Fig. 15, assuming no further N transformations below the 55 cm depth sampled, mound systems show extensive denitrification in 3 of the 4 depths sampled. The average NO_3-N level in the soil solution in the sand fill is 44.5 mg/L and 5 cm into the natural soil it is 64.5 mg/L, apparently a 45% increase. However, since the same area shows a 61.5% increase in Cl^-, there must in fact already be a small loss (approx. 17%) of NO_3-N due to denitrification in this area.

The average NO_3-N concentration is reduced to 19.5 mg/L at the 55 cm depth, apparently a net loss of 69.8%. However, by comparing NO_3-N, TKN and Cl^- levels, more accurate assessments of the actual changes in the soil become possible. By comparing the NO_3-N and Cl^- levels at 5 and 15 cm depths, it emerges that, although there seems to be a loss of NO_3-N in this region, in fact, on the average, there is a 4% rise in

NO_3-N. Moreover, because the TKN does not fall as rapidly as the NO_3-N plus NH_4-N in this region, some loss of NO_3-N should be attributed to immobilization of N by soil microorganisms, rather than to denitrification alone. However, overall, between 5 and 55 cm depth in the soil, the actual percentage reduction of NO_3-N can be set at about 31% (approx. 70% reduction in NO_3-N minus 39% reduction in Cl^-). Combining the actual NO_3-N gains and losses at the soil surface (i.e. to 5 cm depth) and in the natural soil (i.e. from 5 to 55 cm depth) an overall average reduction of 44% in NO_3-N (17 - 4 + 31 = 44%) can be attributed to denitrification in mound systems. As now functioning, mound systems thus destroy 44% - 33% = 11% more of the fixed N leaving the system than was projected on the base of laboratory work with soil columns (Magdoff and Keeney, 1976). Under proper dosing regimes, this figure could be even higher.

However, if through heavy dosing a mound system develops a crust at the gravel/sand interface, it may lose the saturated zone at the natural soil surface and inputs of fresh carbon from effluent surges. As regards N transformations, this mound would then function more like a conventional system, reducing its N loss due to denitrification. The issue must therefore be addressed whether mounds as operating now or in the future pose a serious threat to public health by nitrate contamination of groundwater. This question can be examined by comparing the situation with mounds to experiences with conventional septic sysems, where nitrate movement below and around seepage areas and its effect on groundwater quality have been repeatedly studied.

Pruel (1967) found averages of <1 mg/L NH_4-N and <10 mg/L NO_3-N in groundwater samples collected at distances of about 50 feet from 6 conventional septic systems in sandy soils in Minnesota; the septic tank effluent in these systems contained 40-60 mg/L NH_4-N, i.e. values in the same range as averages for the 33 mound systems studied in Wisconsin (Table 9). However, occasionally NO_3-N in excess of 10 mg/L was found in groundwater 90 feet from some systems. Since the recommended minimum setback between a domestic well and a septic system seepage area in Wisconsin is only 50 feet, this suggests that elevated NO_3-N might sometimes be encountered in groundwater near conventional systems or crusted mound systems with loading and site characteristics similar to the older, presumably crusted systems examined by Pruel (1967).

In septic system seepage collected over 2 years from lysimeters in 3 different soils (sand loam, sandy clay, clay), only samples collected one summer from the sandy loam showed mean NO_3-N values above 10 mg/L (Brown et al., 1977, 1978); maximum values of 40 mg/L occurred 100 cm away from the seepage trench even though the average TKN in the septic tank effluent was only 29.8 ± 11 mg/L (versus 81.5 ± 5.2 mg/L TKN for septic tank effluent in 33 mound systems; Table 9). On the basis of nitrogen flux calculations made using data from this lysimeter work, a maximum density of 1 conventional system per 4.94 acres (or 50 systems per km^2) was recommended for the best situation in the central southeastern U.S.A. (Brown et al., 1978). Similarly, depending on soil hydrological conditions, densities of only 1-10 systems per km^2 were deemed permissible for northern and southern Wisconsin, while septic systems were not recommended at all for the bulk of the state for fear of groundwater contamination by nitrate.

However, in general such caution cannot be justified in practice. In studies of 4 operating conventional septic systems in Wisconsin (Walker, 1973; Walker et al., 1973a,b), samples of soil collected below crusted, ponded seepage beds contained an average of 83.8 mg/L NO_3-N in the soil solution (range 20-160 mg/L); the septic tank effluent entering the systems contained an average of 66.25 mg/L NH_4-N and 80 mg/L TKN (values in the same range as those found for 33 mound systems; Table 9). Thus, it was assumed that all the NH_4-N entering such systems was being nitrified below the crust and might cause contamination of groundwater. Maximum levels of NO_3-N in the uppermost 30 cm of groundwater under a heavily loaded system (6.0 cm/day = 1.44 gal/ft^2/day) were 40 mg/L and still 15 mg/L 100 feet downgradient from the seepage area; however, the corresponding mean NO_3-N levels were only 29 and 9 mg/L. There was no NO_3-N contamination upgradient in the groundwater flow. Perched groundwater below another system showed 9-28 mg/L NO_3-N in wells less than 10 feet away from the system but only traces to 1 mg/L NO_3-N occurred in the real groundwater below a clay layer or in the perched groundwater 25 feet away. Average values of 7.3 mg/L NO_3-N were found below a 3-fingered trench system (maximum 16 mg/L), but most individual and all average levels in samples collected downgradient in the water table were below 10 mg/L. Little nitrification took place in a fourth system studied because its seepage trench was located in anaerobic groundwater. The systems monitored by Walker et al. (1973a,b) were the worst systems encountered from the N flux standpoint in previous studies by Bouma et al. (1972). Groundwater NO_3-N above the 10 mg/L level was not encountered near the other conventional systems examined by Bouma et al. (1972) nor in wells close to 4 experimental mound systems they examined. Similarily, no NO_3-N levels in excess of 10 mg/L were found in high groundwater near 4 commercially installed mound systems now studied (cf. p. 154 and Table 21). The mean concentration of NO_3-N in groundwater adjacent to 11 conventional septic systems in another study in Wisconsin ranged from 7-40 mg/L beside the seepage trenches or drywells to 1-6 mg/L 45 feet downgradient (Dudley and Stephenson, 1973). Again no contamination was encountered upgradient. The highest values observed (65-85 mg/L) occurred in the spring and fall alongside one drywell, but most values rarely exceeded 20 mg/L alongside any other seepage area. Similar field studies of 9 systems in Ontario (average effluent NH_4-N = 63.0, TKN = 75.1 mg/L) also revealed highest groundwater NO_3-N next to the system (range 0-33 mg/L) but usually safe (<10 mg/L) values 50 feet downgradient in the water table (Chan, 1978). Only in wells located close to 3 seepage areas serving 1 house and 4 seasonally occupied cottages were unsafe NO_3-N levels observed beyond 50 feet (range 0.5-19 mg/L; average 8.76 mg/L). NO_3-N levels above 10 mg/L were not encountered 40 feet from the seepage areas in similar studies conducted earlier with one large system in Wisconsin (Polkowski and Boyle, 1970) and under only 1 of 19 systems studied in Michigan (Ellis and Childs, 1973).

Thus, wherever nitrate pollution of groundwater by a septic system has been observed in the field, seriously elevated NO_3-N levels have been encountered only occasionally in the upper levels of the groundwater in the immediate vicinity of the seepage area or nearby downgradient from it in the water table. In studies of groundwater nitrate levels in Wisconsin (Schuknecht et al., 1975; Delfino, 1977), Minnesota (Woodward et al., 1961), Missouri (Keller and Smith, 1967), Illinois (Walker, 1973), California (Behnke and Haskell, 1968), Kansas (Riddel et al., 1974), and Oregon (Quan et al., 1974), rarely could septic systems be clearly implicated as sources of excessive nitrate pollution; other point or diffuse sources associated with agricultural practices (feedlots, barnyards, fertilized fields) were more frequently suspected causes of elevated NO_3-N levels in well water.

Since NO_3-N enrichment in groundwater by percolating water is usually encountered only in the uppermost layer, i.e. no intermixing with lower layers is observed, the simple expedient of using deeply cased wells to tap only deeper groundwater for drinking affords good protection from any possible nitrate or bacterial contamination from wastewater disposal seepage (Delfino, 1977). Wells in Wisconsin are usually cased to a minimum depth of about 30 feet into the water table in water-bearing sands or 70 feet in creviced bedrock (C. Goethel, Wis. Dept. of Natural Resources, personal communication).

High nitrate levels in water (Winton et al., 1971) or in foodstuffs (Maynard et al., 1976) seriously affect only infants up to an age of 3-6 months (Gruener and Shuval, 1970; Kriese, 1974). Vitamin C, e.g. in orange juice, apparently acts as an antidote against and confers tolerance toward nitrate (nitrite) poisoning (Shuval and Cruener, 1972; Kriese, 1974) so that methemoglobinemia is now rarely observed under current water supply and dietary practices. Thus, both conventional septic systems and mound systems, whether operating as now with better nitrate removal than conventional systems (cf. p. 95) or not, can undoubtedly be tolerated at reasonable densities without any serious public health hazard from nitrate in drinking water. Obviously if shallow wells are to provide good quality, low-nitrate drinking water, they should not be placed closer than 100 feet downstream in the water table from septic system soil absorption area. Anyone concerned about the quality of their drinking water should have it tested.

Phosphorus movement

There is no direct public health hazard that can be associated with P movement from septic systems; the only concern is an environmental issue, viz. the potential for enhancement of eutrophication of surface waters by recharge with P-laden groundwater. P is the element that limits growth of algae, especially nuisance blue-green algae, in most lakes in the temperate zones of the northern hemisphere (Shannon and Brezonik, 1972; Schindler, 1977). The issue therefore becomes whether septic systems in general and mound systems in particular are potentially significant sources of P pollution to surface waters.

Normally soil exhibits an amazingly high capacity for immobilization of P through ion-exchange, sorption and precipitation reactions. P applied in fertilizers (Kao and Blanchar, 1973) or in sewage water (Beek et al., 1977; Hill, 1972) is completely retained even in sands (Tofflemire and Chen, 1977; Aulenbach, 1977) as water percolates through soil. Incomplete P removal occurs only under conditions of saturated flow (Viraraghavan and Warnock, 1976a,b; Dudley and Stephenson, 1973; Brandes et al., 1975; Sawhney, 1977; Magdoff et al., 1974).

Presently many mound systems are being overdosed; as a result saturated conditions prevail at least temporarily in a majority of the soils below the mounds. This condition is probably responsible for the high OP levels found below several mounds. The average value of 0.9 g OP/g of dry soil at the 55 cm depth in the natural soil is about 2-10 times the level found by Brown et al. (1978) in sand loam, sand clay, and clay in lysimeters under conventional seepage fields. On the soil solution basis, the average OP under mound systems would thus seem to be about 5-10 times higher than OP found in groundwater samples collected in wells around numerous conventional systems: 4 systems (Walker, 1973), 7 systems (Bouma et al., 1972), 9 systems (Chan, 1978), and 1 system each (Polkowski and Boyle, 1970: Viraraghavan and Warnock, 1967a,b; Jones and Lee, 1977; Peavy and Groves, 1977). Thus, as regards P movement into the soil, the mounds are apparently not as good as conventional systems. If dose volumes are adjusted as specified in the design criteria to maintain an unsaturated condition in the natural soil, mounds should function as well as or better than conventional systems with respect to P removal.

Despite the high levels of OP and TP in the fill and soil underneath the mounds, there appears to be very little net movement of P into groundwater, as chemical analysis of groundwater samples has revealed, as discussed below.

Since all recent evidence indicates that P movement from septic systems is minor or negligible (Walker, 1973; Bouma et al., 1972; Jones and Lee, 1977; Peavy and Groves, 1977; Dudley and Stephenson, 1973;

Ellis and Childs, 1973; McKinney and Krause, 1979) mound systems can therefore be discounted as significant sources of P contributing to surface water eutrophication.

Groundwater quality

To establish whether mound system seepage could contribute to bacterial contamination or nutrient enrichment of surface waters through transfer via shallow aquifers, suitable systems from the 33 under surveillance were selected for closer examination of nearby groundwater quality.

Four systems (3-47, 3-72, 3-78, and 3-82) were located near lakes over shallow permanent water tables that could easily be sampled using the PVC wells described earlier (p. 45). Wells were therefore placed near these systems in locations intended to intercept flow of effluent away from the mounds; in a few instances the direct flow could not be intercepted due to site limitations (trees, landscaping, etc.) or because homeowners wished the wells placed so as to minimize inconvenience to them.

System 3-47 serves a household on Lake Sugarbush in Vilas County, Wisconsin. The residence is occupied year around (1 occupant during the week, 3-7 on weekends). The mound and location of the groundwater wells are shown in Fig. 28; the main direction of groundwater flow is indicated by the large arrow. The soil is an Alcona fine sandy loam. Wells 1 and 4 are located at the toe of the mound and 12'6" and 10' respectively from the absorption bed. Wells 2 and 5 are 20 ft from the absorption bed; Well 3 is 35 ft from the absorption bed and 35 ft from Lake Sugarbush. All wells were sunk deep enough to sample at least the

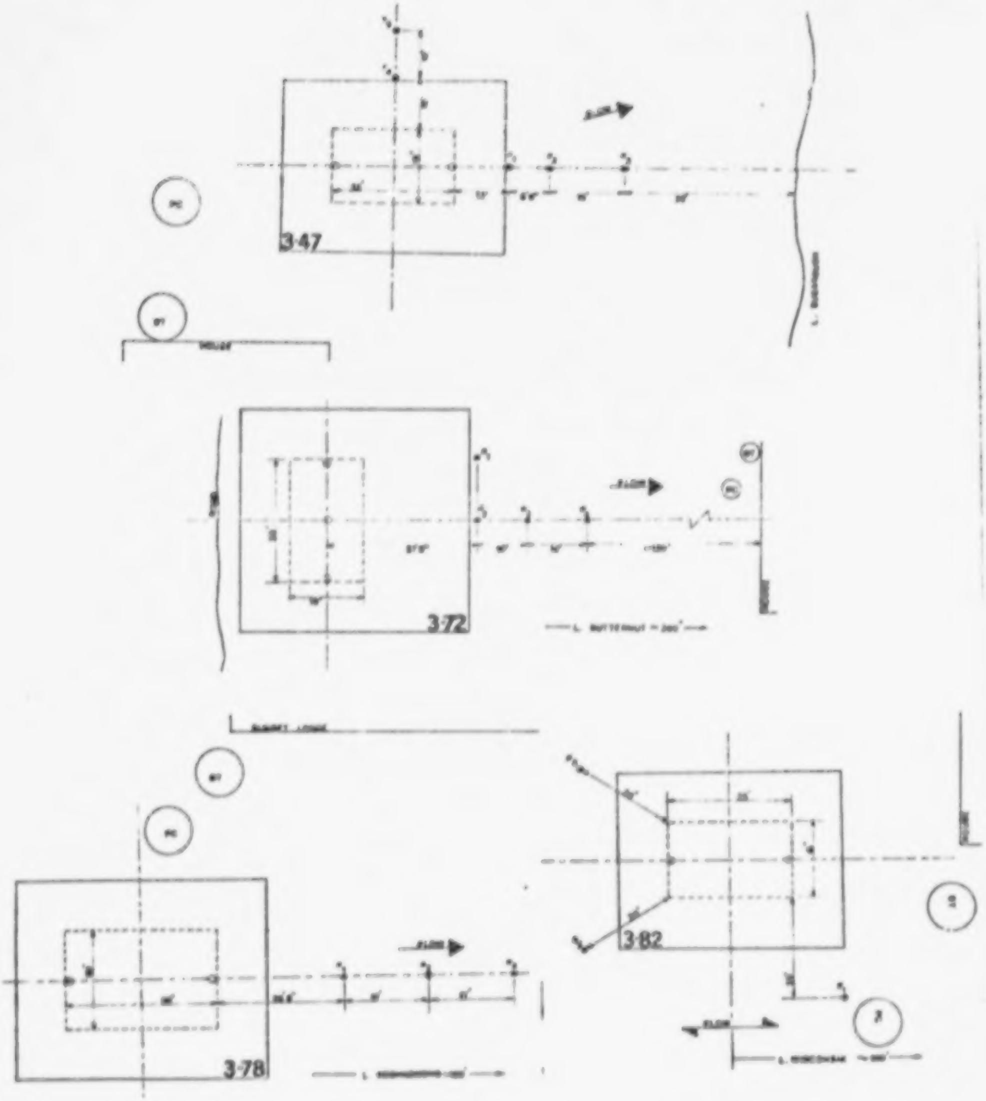
3·47
3·72
3·78
3·82
FLOW
FLOW
FLOW
FLOW
HOUSE

Fig. 28. Location of the groundwater wells, direction of groundwater flow and absorption area for systems 3-47, 3-72, 3-78 and 3-82. ST, septic tank; PC, pumping chamber.

upper 30 cm of the groundwater as it existed on 5/22/78 when the wells were installed. The direction of groundwater flow was determined using the water temperature, electrical conductivity, and chloride levels (Gibbs, 1977).

Table 21 shows the results of analysis of water samples obtained from each well during the spring and summer of 1978. The anaerobic nature of the groundwater was indicated by the precipitation of iron oxides in samples soon after they were brought to the surface. Because of personnel changes in the bacteriological testing laboratory which provided bacterial analyses on the water samples collected, some tests on groundwater samples were not performed (indicated by hyphens in Table 18) while others were performed without an appropriate degree of sensitivity (samples filtered for assays by the membrane filter technique were too small). Counts reported as <100, for example, thus indicate insufficient sensitivity in the bacteriological assay, not high bacterial counts in the samples tested. Anomalously high FS, and *Ps. a.* values obtained from samples taken from Well 1 on 7/10/78 may be due to a neighborhood dog which removed the well cap and urinated on the well (personal communication with homeowner).

Tests performed with sufficiently large samples showed that the groundwater usually was of a bacteriological quality as good as or close to drinking water quality. Discounting the bacterial values in samples from Well 1 on the day of dog interference, the bacterial levels are all at or below background levels; thus few if any bacteria travel from the mound. All TC, FC and TS counts are as low as or lower than those encountered in samples taken from groundwater wells protected with nylon

Table 21.

Results of bacterial and chemical analysis of water samples from pumping chambers of and from groundwater wells near systems 3-67, 3-72, 3-78 and 3-82, as well as from nearby lakes (1, 2, 3, 4, 5 represent Wells 1, 2, 3, 4, and 5; E is effluent sample; L is lake sample)

Date and source		MPN/100 ml				mg/L				
		TC	FC	FS	*Ps. a.*	TKN	NH_4-N/NO_3-N	TP	OP	Cl^-
System 3-67										
5/22/78	E	5.4×10^6	3.8×10^5	2.1×10^5	2.2×10^3	69.88	76.6/0.2	19.5	11.0	92.1
	1	<2	<2	0	0	1.54	0/0	0.2	0.15	84
	2	<2	<2	0	0	0.44	0/0	0	0	59.7
	3	<2	<2	0	0	1.65	0/0	1.0	0.4	8.0
	4	<2	<2	0	0	0.99	0/0	0	0	12.8
	5	<2	<2	0	0	1.54	0/0	0	0	6.6
	L	-	-	-	-	0.55	0.1/0	1.8	1.5	6.7
6/12/78	E	3.5×10^5	7×10^4	1.0×10^6	5×10^2	72.35	64.5/0	17.5	17.5	-
	1	7	4	<100	17	0.66	0/0	0.61	0.25	11.2
	2	12	<2	<100	<2	0.66	0/0	0	0	7.8
	3	2	<2	<100	<2	0.65	0/0	0	0	8.2
	4	6	<2	<100	<2	0.77	0/0	0	0	11.8
	5	8	<2	<100	<2	0.33	0/0	0	0.2	6.2
	L	-	-	-	-	0.88	0/0	1.15	0.35	6.8
7/10/78	E	2.2×10^6	7.9×10^5	2.2×10^5	5.4×10^4	71.1	70.1/0	18.0	11.0	-
	1	<20	<20	200	5.4×10^3	1.21	0/0	1.25	1.18	-
	2	<20	<20	<100	<20	0.88	0/0	0	0	9.8
	3	<20	<20	<100	<20	0.66	0/0	0	0	7.8
	4	<20	<20	<100	<20	0.66	0/0	0	0	-
	5	<20	<20	<100	<20	0.35	0/0	0	0	6.8
8/20/78	E	2.44×10^7	1.7×10^6	1.8×10^5	2×10^2	63.12	64.6/2.2	62.0	31.2	63.9
	1	4	2	<100	<2	1.43	0.3/16.3	1.8	1.5	25.8
	2	<2	<2	<100	<2	0.88	0.2/0.1	0	0	8.0
	3	<2	<2	<100	<2	0.44	0.7/0.05	0	0	12.5
	4	11	<2	<100	<2	0.44	0/0.05	0	0	6.6
	5	<2	<2	<100	<2	0.44	0/0	0	0	6.6
	L	-	-	-	-	1.65	0/.79	0.25	0.25	6.0
7/22/78	E	-	-	-	-	71.65	50.4/0.2	6.0	4.5	82.3
	1	-	-	-	-	2.64	0.4/0.6	0.28	0.2	41.6
	2	-	-	-	-	0.44	0.05/0.3	0	0	23.6
	3	-	-	-	-	0.22	0/0.1	0	0	11.2
	4	-	-	-	-	0.11	0/0.1	0	0	45.1
	5	-	-	-	-	0.0	0/0	0	0	25.6
System 3-72										
5/22/78	E	3.5×10^6	1.7×10^6	5.9×10^4	1.4×10^3	56.4	51.1/0.5	68.0	41.0	61.2
	1	200	<2	0	0	0.77	0/0	0.25	0.15	41.0
	2	<2	<2	0	0	0.66	0/1.9	0.20	0.15	53.4
	3	<2	<2	0	0	5.07	0/0	0	0	39.4
	4	<2	<2	0	0	1.1	0/0	0	0	35.9
	L	-	-	-	-	0.88	0/0	0	0	7.4
6/12/78	E	9.2×10^6	9.2×10^6	2.1×10^5	9.4×10^4	63.93	58.6/0	65.1	40.0	60.9
	1	11	2	<100	<2	1.43	0/0	0.15	0.1	42.3
	2	33	7	<100	<2	0.55	0/3.4	0	0	57.4
	3	10	8	<100	<2	0.33	0/0	0.1	0.1	38.4
	4	2	2	<100	<2	0.44	0/0	0	0	7.3
	L	-	-	-	-	0.0	0/0	0	0	7.4
7/10/78	E	2.4×10^7	3.5×10^6	2.2×10^5	2×10^2	70.9	72.8/1.0	47.3	41.0	63.4
	1	80	20	<100	<20	5.5	0/0.2	0	0	47.0
	2	790	790	50	<20	0.66	0/0	0	0	59.6
	3	<20	<20	<100	<20	0.11	0/0	0	0	38.9
	4	<20	<20	<100	<20	0.0	0/0	0	0	7.6
	L	-	-	-	-	0.0	0/0	0.1	0.1	7.0
7/22/78	E	-	-	-	-	70.2	61.9/0.9	75.0	62.0	62.3
	1	-	-	-	-	0.44	0/0.2	0.15	0.1	73.0
	2	-	-	-	-	0.22	0.05/0	0	0	80.4
	3	-	-	-	-	0.44	0/0.6	0	0	55.6
	4	-	-	-	-	0.44	0.2/0.4	0	0	15.0
8/25/78	E	-	-	-	-	80.69	60.6/0.9	63.4	38.1	67.3
	1	<2	<2	<100	<2	1.65	0.5/0.3	0.2	0.15	59.1
	2	<2	<2	<100	-	0.99	1.0/0.05	0	0	61.4
	3	-	-	-	-	0.77	0.7/0.3	0	0	38.8
	4	-	-	-	-	1.10	1.3/0.05	0	0	6.8

Table 21

(continued)

Date and source		MPN/100 ml				mg/L				
		TC	FC	FS	Ps. a.	TOC	NH_4-N/NO_3-N	TP	OP	Cl^-
System 3-75										
6/19/78	E	-	-	-	-	52.03	-	-21.0	17.2	91.3
	1	-	-	-	-	2.31	-	0.4	0.3	72.0
	2	-	-	-	-	1.98	-	0.4	0.4	60.2
	3	-	-	-	-	0.99	-	0.35	0.3	42.0
5/10/78	E	-	-	-	-	126.7	120.2/0	27.0	21.0	97.3
	1	-	15	2	<2	0.99	0/0	0.1	0.02	35.4
	2	2.4×10^{3}	10	0	<2	0.88	0/0	0	0	32.4
	3	2.4×10^{3}	2	0	0	0.11	0/0	0.1	0.05	20.0
6/6/78	E	2.4×10^{7}	2.4×10^{7}	9.3×10^{6}	-	120.61	139.2/07	35.0	31.0	122.6
	1	920	<2	<100	<20	0.99	0/0	0.15	0.10	23.0
	2	540	<2	<100	<20	1.10	0/0	0	0	28.9
	3	5	<2	<100	<20	0.66	0/0.5	0	0	-
	L	-	-	-	-	5.96	0/0	0.35	0.32	31.0
6/22/78	E	-	-	-	-	136.93	130.2/0.2	18.0	16.0	-
	1	-	-	-	-	1.21	0/0	0	0	-
	2	-	-	-	-	1.43	0/0	0	0	-
	3	-	-	-	-	0.33	0/0	0	0	28.6
	L	-	-	-	-	2.97	0/0	0.18	0.10	33.9
6/27/78	E	5.4×10^{6}	6.5×10^{6}	6.7×10^{2}	6.2×10^{2}	66.61	94.6/6.9	12.0	10.0	100.3
	1	-	1.1×10^{6}	-	-	2.31	0/0	0.1	0.1	26.1
	2	1165	93	190	370	1.76	0/0.8	0.15	0.05	67.6
	3	163	6	20	<100	2.09	0/0	0.10	0.1	79.4
7/19/78	E	1.6×10^{6}	1.6×10^{6}	1.13×10^{5}	200	100.97	113.5/0.9	11.2	9.6	116.6
	1	3.5×10^{3}	3.5×10^{3}	1.96×10^{4}	<20	0.88	0/0.3	0.07	0.05	27.6
	2	<20	<20	<100	<20	2.77	0/0.5	0.15	0.1	119.1
	3	<20	<20	<100	<20	1.65	0/0	0.2	0.13	67.3
	L	-	-	-	-	0.66	0.22/0.57	0.3	0.1	62.6
8/15/78	E	2.4×10^{2}	9.5×10^{6}	1.7×10^{6}	<200	124.3	-	13.1	10.0	99.1
	1	3.2×10^{3}	100	5.8×10^{3}	<200	1.43	-	0	0	63.9
	2	2.5	<1	<1	<2	1.32	-	0	0.05	26.2
	3	1.6	<1	<1	4	1.21	-	0	0.05	17.7
	L	-	-	-	-	1.98	-	9.6	0.025	27.0
System 3-02										
6/19/78	E	-	-	-	-	61.4	-	28.0	27.5	255.2
	1	-	-	-	-	6.18	-	0	0	270.6
	2	-	-	-	-	3.52	-	0	0	56.8
	3	-	-	-	-	3.61	-	0	0	62.6
5/10/78	E	-	-	-	-	35.66	37.6/0.1	26.2	26.2	-
	1	63	-	7	13	1.31	0/0	0.65	0.65	63.4
	2	1.3×10^{4}	-	0	36	1.76	0/0	0.15	0.05	90.4
	3	63	-	0	7	6.07	2.9/0	0	0.02	307.3
6/12/78	E	3.5×10^{6}	1.3×10^{6}	1.4×10^{5}	270	30.07	35.0/0	66.0	65.2	78.5
	1	33	33	<100	<20	6.07	1.2/0	0	0	360.4
	2	<20	7	<100	<20	1.43	0/0	0	0	21.4
	3	120	16	<100	<20	0.66	0/0	0	0	21.1
	L	-	-	-	-	2.09	0/0	0	0	16.2
6/22/78	E	1.4×10^{6}	2.0×10^{3}	-	-	67.02	37.4/0	63.0	62.0	68.1
	1	38	-	-	-	80.3	1.4/0	0	0	331.9
	2	140	-	-	-	1.21	0/0	0	0	26.0
	3	-	-	-	-	0.66	0/0	0	0	17.4
7/19/78	E	5.4×10^{6}	1.7×10^{6}	2.0×10^{5}	200	62.21	50.6/1.3	51.0	63.2	610.9
	1	20	20	100	20	5.51	3.0/0.1	0	0	606.7
	2	<20	<20	100	<20	3.76	0.05/0.2	0	0	50.1
	3	<20	<20	100	0	0.66	0/0	0	0	69.2
	L	240	79	1.25×10^{4}	<2		0.11/0.4	0.1	0.02	61.3
8/2/78	E	-	-	-	-	-	23.9/0	15.2	10.5	-
	1	-	-	-	-	-	2.2/0/.2	0	0	-
	2	-	-	-	-	-	0.6/0	0	0.02	-
	3	-	-	-	-	-	0/0.3	0	0	-
8/15/78	E	3.9×10^{6}	8.3×10^{5}	2.2×10^{5}	200	66.25	23.6/8.8	37.1	32.5	269.5
	1	10.7	<1	<100	<2	3.52	2.3/0.21	0	0	260.1
	2	1.46×10^{4}	653.3	<100	<2	0.99	0.3/0.1	0	0	21.3
	3	550	66	<100	<2	0.77	0/0	0	0	23.0
	L	-	-	-	-	3.14	-	1.15	0.04	13.9
6/20/79	E	4.7×10^{6}	2.1×10^{6}	2.9×10^{5}	-	-	-	-	-	-
	1	<1,<1	<1,<1	<1,<1	-	-	-	-	-	-
	2	<1,<1	<10,<1	<1,<1	-	-	-	-	-	-
	3	<1,<1	<10,<1	<1,<1	-	-	-	-	-	-

mesh casing installed in areas of high groundwater near 9 conventional systems (Chan, 1978). Since the counts in this study were below those permissible by Canadian Public Surface Water Supply criteria, bacterial contamination was considered to be not significant.

From the standpoint of nutrients, only well 1 on 8/20/78 showed levels exceeding typical background levels of either NO_3-N or OP. On 8/19/79, the area received 3.2" of rain (personal communication from homeowner); the high 8/20/78 levels could be the result of the precipitation flushing the system and surrounding terrain and contaminating Well 1. Normally well samples did not contain any detectable nitrate. Since the groundwater is anaerobic, if small amounts of nitrate are leached into it, they are presumable slowly denitrified under normal circumstances. Heavier NO_3-N loads leached into the groundwater by the heavy rains on 8/19/79 had apparently not had a chance to dissipate before the 8/20/79 samples were taken.

If it is assumed that septic tank seepage is the only major source of Cl^- in the vicinity of the mound, the levels of Cl^- in the well water samples give a good indication of the extent of dilution that the mound seepage has undergone by the time that the samples were taken from the wells. Dilution ratios can be calculated on the basis of the Cl^- levels in the septic tank effluent by assuming that mound seepage is diluted with groundwater containing 6.0-6.8 mg/L of Cl^-, the background Cl^- level in the lake water. For example, using data from the 5/22/78 sampling for mound 3-47, 1 L (1000 ml) of septic tank effluent with 92.1 mg/L of Cl^- need be diluted with only 104.8 ml of groundwater with 6.7 mg/L to give water with the 84 mg/L of Cl^- found in Well 1, but with 13

L of such groundwater to give water with the 12.8 mg/L of Cl^- found in Well 4. However, since the bacteria and nutrients are attenuated by factors much higher than these dilutions, obviously the mound is removing the bulk of the bacteria and nutrients before seepage reaches the groundwater. In summary, 3-47 shows little if any effect on the groundwater quality below the system.

System 3-72 serves a house with 2 full-time residents and many visitors on weekends; it is located on the shores of Lake Butternut in Price County, Wisconsin. The soil is a Pence loam overlain by 3-4 feet of various sand fills. Well locations and the main direction of groundwater flow at the site are indicated in Fig. 28. Well 1 is an old well installed prior to the development of the Typar-coated wells (cf. p. 17). Results of water sample analysis are shown in Table 21. As for system 3-47, there is no significant bacterial movement away from this system. Again the chloride data indicate that the mound seepage has not been signficantly diluted in most of the well water samples collected. In samples taken from Wells 1 and 2 on 7/22/78, the Cl^- was even higher than that in the septic tank. This may indicate a front of higher Cl^- passing through the area of the wells on that day, or supplementation of the Cl^- in the groundwater by Cl^- from another source, e.g. lawn fertilizer. In any case, the nutrient values are attenuated by factors much greater than the dilution of the seepage. Thus, no significant effect of this system on the quality of the groundwater can be shown.

System 3-78 serves a bar located on the north shore of Lake Koshkonong in Jefferson County, Wisconsin. The native soil is a Keowns silt loam. The locations of the wells and the direction of groundwater flow are shown in Fig. 28. This bar attracts few patrons during the week but many on the weekends. Chemical contamination of the groundwater by system 3-78 is minimal with all levels close or equal to normal groundwater levels (Table 21). The reliable bacterial analyses obtained indicated no bacterial contamination of the groundwater. The high counts recorded for some sample from Well 1 are suspected to be artifacts due to inappropriate techniques being used in the bacteriological laboratory at the time.

System 3-82 serves a household with 3 full-time residents who receive many visitors on weekends. It is located on the shores of Lake Wisconsin in Columbia County, Wisconsin. The location of the wells and direction of the groundwater flow are shown in Fig. 28. The groundwater depth in the Plainfield loamy sand changed constantly during the sampling period due to changes in the level of Lake Wisconsin. After the homeowner installed a dog house next to Well 2, higher TC and FC levels were observed in some samples from that well. However, these higher counts may have been artifacts due to inappropriate laboratory techniques. The 6/20/79 samples from all wells were analysed in duplicate by the Wisconsin State Laboratory of Hygiene and indicate that no bacterial contamination of the groundwater with seepage from the mound is occurring. No significant chemical contamination of the groundwater by this system is indicated (Table 21).

In conclusion, none of the four systems monitored closely for groundwater effects show any contamination that could be associated with significant transfer of bacteria or nutrients from mound seepage to the groundwater. All 4 systems performed better in this respect than conventional systems studied in Canada (Chan, 1978).

Prospects for Continued Satisfactory Performance

At present, practically all of the mound systems in Wisconsin are apparently performing satisfactorily, even though some have not been installed and are not being operated exactly as specified in the design recommendations. Seventeen of 20 mound systems studied which furnished reliable flow data had dose volumes higher than those specified in the original package designs; most of the mounds had fill material with a higher uniformity coefficient than that of the fill type recommended (medium sand). Nevertheless, Wisconsin mounds still perform satisfactory bacterial purification and reduce nitrate inputs to groundwater better than typical conventional subsurface wastewater disposal fields. System 3-64 is an excellent example of a mound system with features that depart significantly from the design recommendations; it had a high dose volume (569 gallons), a high dosing rate (1.05 gal/ft^2/dose), a high loading rate (1.16 gal/day/ft^2), and a fill with texture of a loamy sand (UC = 9.9, 15.7% fines). Despite these shortcomings at no time during the two-year monitoring program did this system allow surface discharge of untreated effluent, and it achieved good bacterial and chemical renovation of the wastewater.

Three mound systems that failed during the winter months (1-160, 3-110, 3-112) all failed due to improper construction, not faulty design. System 1-160 had little or no insulating cap or cover, and both 3-110 and 3-112 had inappropriate fill material with an average uniformity coefficient of 50 and 19.5% fines.

Mound systems that are properly installed and which are dosed in accordance with design guidelines purify septic tank effluent better than mounds which are being overdosed, and much better than conventional seepage systems. All mounds perform as well as or better than conventional systems and are expected to continue to operate satisfactorily for periods longer than conventional systems, since their dosing system retards or eliminates the formation of a clogging mat below the gravel in the seepage area.

If in time a clogging mat or crust should form and become so resistant to septic tank effluent infiltration as to interfere with mound performance, it can easily be removed by chemical treatment using hydrogen peroxide according to the POROX process (Harkin et al., 1975).

Recommendations

Most mound systems are presently being incorrectly dosed (dose volumes too large, dosing frequency too low). In such cases, the float switches in the pumping chamber should be adjusted and the dose volume calibrated to bring the mound loadings into compliance with the original package designs. The Wisconsin Division of Health is considering reducing the size of the pumping chamber to force the reduction of dose volumes (personal communication from David Frederickson, 1979). However this strategy would reduce the storage capacity of the chambers in case of pump failure. It would be better to specify in plan approvals what dose volumes must be used and to insist that dose volumes be calibrated before the mound is allowed to operate. If the dose volume is controlled by limit switches, these should be anchored onto the sides of the pumping chamber to prevent the homeowner from tampering with the levels and changing the size of the dose volume.

Laboratory work has revealed that the maximum loss of fixed nitrogen by nitrification/denitrification and fastest breakdown of organic matter occur when a system cycles from aerobic to anaerobic conditions once every 6-12 hours (Patrick and Reddy, 1974, 1976). Thus, dose volumes should be calibrated to dose two to four times daily to minimize the nitrate entering groundwater as well as to promote the unsaturated flow in the fill material that will improve bacterial attenuation and P removal, and delay clogging.From this standpoint it would be preferable to trigger a dosing cycle four times a day by use of an automatic timer; limit switches allow larger doses to build up in the pumping chamber which are transferred to the mound only at much less frequent intervals in systems not receiving large daily wastewater inputs.

The fill material is of prime importance in the effectiveness of operation of the mound systems. Plan approvals should specify that a particle size analysis of the fill material be performed prior to the placement of the fill. The original package designs states that "the fill material need not be washed as a certain content of fines improves the filtering capacity of the fill material." This statement creates a loophole for use of a fill that is too heterogeneous. Too coarse or fine a fill material, fosters bacterial movement or may lead to surface seepage.

The Wisconsin mound designs use a pressurized distribution system, intended to guarantee even distribution of effluent over the absorption area. When used to apply small doses frequently, this system assures unsaturated flow, prevents the formation of a crust, and promotes bacterial and chemical purification of the effluent in the fill. No mound examined had developed a mature crust. It appears that even with excessive dose volumes the pressurized distribution network is delaying soil clogging and crust formation. No problems due to pump failures were reported during the past 3 years; commercial pumps are therefore capable of handling septic tank effluent in mound systems without problems. Consideration should be given to incorporating pressurized distribution systems into conventional soil absorption fields.

Because cold temperatures apparently contributed to the failure of three improperly installed systems, the importance of an adequate cap or cover on the mound systems to provide thermal insulation must be emphasized in the package designs. In addition, the pumping chamber should be insulated to prevent cooling of the effluent in the chamber. Several layers of styrofoam building insulation underneath the lid would help keep the water in the pumping chamber warm.

At present the mound systems are being over-designed by assuming a household water use of 150 gal/bedroom/day. The actual average water use was determined to be 43.5 gal/person/day or approximately 87 gal/day/bedroom. These values match typical average water use figures determined for rural homes by many investigators and suggest that the size requirements for the mounds can be reduced. However, water use in some households may vary dramatically from the average volumes (e.g. 114 gal/person/day for system 3-64). Consequently the large safety factor built into the sizing of mounds may be necessary. Households served by mounds should be instructed (e.g. with extension brochures or bulletins) that excessive water use may lead to malfunction or failure of their wastewater disposal system. Owners should be advised that faulty plumbing (leaking faucets, running toilets, etc.) must be corrected promptly and that living habits that lead to massive water use (frequent prolonged showers, excessive laundering, excessive use of water softeners, etc.) should be discouraged. Use of garbage disposals should be discouraged in households serviced by mound systems, since they overload the septic tank with solids and add unnecessary loads of water to the system especially when these are in a ponded condition. Similarly, intrusion of fill into the gravel of the absorption bed of ponded mound systems could clog the distribution network. Consideration should be given to prescribing better barrier materials between the fill and gravel in mounds than the straw or building paper now specified in H.62.20, Wisconsin Administrative Code. A good barrier material (Harkin, 1978) is fiberglass building insulation or the DuPont Typar® filter fabric incorporated into the construction of the groundwater wells (p. 45).

From the standpoint of technical performance in the retention and treatment of wastewater the Wisconsin mound systems as presently described in the three engineering packages can be highly recommended for use at sites where conventional systems are inappropriate because of problem soils. The soil and site restrictions currently enumerated in the design packages (Converse et al., 1976a-c; Converse, 1978) should be applied as criteria to judge whether a mound system and which type of system is appropriate for any given site. The systems can clearly be installed by commercial contractors who have been trained in good techniques of mound installation as described in the package designs

(Converse et al., 1976a-c; Converse, 1978) provided that adequate review of each installation plan and inspection of the installation during construction are provided by appropriately trained state, county or local regulatory personnel. While these recommendations correspond to current practice, minor shortcomings experienced with the mounds investigated indicate that officials inspecting mound installations must be instructed to ensure that installers follow the package guidelines closely, especially with respect to:

Selection of the appropriate mound type (Package 1, 2, or 3);

Appropriate placement and sizing of the system;

Adherence to the recommendations concerning fill texture;

Adherence to the recommendations concerning the frequency of dosing and the dosing volumes;

Provision of adequate cover for and landscaping of the mound.

LITERATURE CITED

Alexander, M. 1977. Introduction to Soil Microbiology. John Wiley and Sons, Inc., New York. 467 pp.

Aulenbach, D. B. 1977. Chemical interactions in a rapid infiltration system accomplishing tertiary treatment of wastewater. FWI Report 77-5, Rensselaer Fresh Water Institute, Troy, N.Y., 58 pp.

Beek, J., F. A. M. de Haan and W. H. van Riemsdijk. 1977. Phosphates in soils treated with sewage water. J. Environ. Qual. 6:4-12.

Behnke, J. J. and E. E. Haskell. 1968. Ground water nitrate distributions beneath Fresno, Calif. J. Amer. Water Works Assoc. 60:477-480.

Bennett, E. R., K. D. Linstedt and J. T. Felton. 1975. Rural home wastewater characteristics. *In* Proc. Nat. Home Sewage Disposal Symp., Amer. Soc. Agr. Engs., St. Joseph, MI., pp. 74-78.

Bernhard, A. P. 1975. Return of effluent nutrients to the natural cycle through evapotranspiration and subsoil-infiltration of domestic wastewater. *In* Proc. Nat. Home Sewage Disposal Symp., Amer. Soc. Agr. Engs., St. Joseph, MI, pp. 175-181.

Brandes, M., W. A. Chowdry and W. W. Cheng. 1975. Experimental study on removal of pollutants from domestic sewage by underdrained soil filters. *In* Proc. Nat. Home Sewage Disposal Symp., Amer. Soc. Agr. Eng., St. Joseph, MI, pp. 29-36.

Bouma, J. 1975. Unsaturated flow during soil treatment of septic tank effluent. J. Environ. Engr. Div., Amer. Soc. Agr. Engs. 101:967-983.

Bouma, J., W. A. Ziebell, W. G. Walker, P. G. Olcott, E. McCoy and F. D. Hole. 1972. Soil absorption of septic tank effluent--A field study of some major Wisconsin soils. Univ. of Wisconsin Extension Info. Circ. No. 20, Madison, WI, 235 pp.

Bouma, J., J. C. Converse and F. R. Magdoff. 1973. A mound system for disposal of septic tank effluent in shallow soils over creviced bedrock. *In* Proc. Int. Conf. Land for Waste Mangement, Ottawa, Canada, pp. 367-378.

Bouma, J., J. C. Converse, R. J. Otis, W. G. Walker and W. A. Ziebell. 1973. A mound system for on-site disposal of septic tank effluent in slowly permeable soils with seasonally perched water tables. J. Environ. Qual. 413:382-388.

Bouma, J., J. C. Converse and F. Magdoff. 1974a. Dosing and resting to improve soil absorption beds. Trans. Amer. Soc. Agr. Engs. 17:295-298.

Bouma, J., F. Baker and P. Veneman. 1974b. Measurement of water movement in soil pedons above the water table. Univ. of Wisconsin Extension, Geol. Nat. Hist. Surv., Info. Circ. No. 27, Madison, WI, 114 pp.

Bouyoucos, G. J. 1962. Hydrometer method improved for making particle size analysis of soils. Agron. J. 54:464-465.

Bower, C. A. and L. V. Wilcox. 1965. Soluble salts. *In* C. A. Black (ed.), Methods of Soil Analysis. Amer. Soc. Agron., Madison, WI, pp. 168-213.

Bremner, J. M. and D. R. Keeney. 1965. Steam distillation methods for the determination of ammonium, nitrite and nitrate. Anal. Chim. Acta 32:485-495.

Bremner, J. M. and D. R. Keeney. 1966. Kjeldahl total nitrogen determinations in plant tissues. Soil Sci. Soc. Am. Proc. 30:577-582.

Brown, K. W., J. F. Slowey and H. W. Wolf. 1977. The movement of salts, nutrients, fecal coliforms and virus below septic leach fields in three soils. *In* Proc. Second Natl. Home Sewage Disposal Symp., Amer. Soc. Agr. Eng., St. Joseph, MI, pp. 54-66.

Brown, K. W., J. F. Slowey and H. M Wolf. 1978. Accumulation and passage of pollutants in domestic septic tank disposal fields. Texas A&M Research Foundation, College Station, TX, 125 pp.

Butler, R. G., G. T. Orlab and P. J. McGauhey. 1954. Underground movement of bacterial and chemical pollutants. J. Amer. Water Works Assoc. 46:97-111.

Chan, H. T. 1978. Contamination of the Great Lakes by private wastes. International Joint Commission, Ontario, Canada, 191 pp.

Clausen, E. M., B. L. Green and W. Litsky. 1976. Fecal streptococci indicators of pollution. *In* D. Hoadley and W. Dutka (ed.), Bacterial indicators/health hazards associated with water. Amer. Soc. Testing Mats. Philadelphia, PA, Publ. No. STP635, pp. 247-264.

Cliver, D. O. 1975. Virus association with wastewater solids. Environ. Letts. 10:215-223.

Comly, H. H. 1945. Cyanosis in infants caused by nitrates in well water. J. Amer. Med. Assoc. 129:112-116.

Converse, J. C. 1978. Design and construction manual for Wisconsin mounds. Small Scale Waste Management Project, Univ. of Wisconsin, Madison, WI, 84 pp.

Converse, J. C., R. J. Otis and J. Bouma. 1976a. Design and construction procedures for fill systems in slowly permeable soils with or without high water tables. Small Scale Waste Management Project, Univ. of Wisconsin, Madison, WI, 24 pp.

Converse, J. C., R. J. Otis and J. Bouma. 1976b. Design and construction procedures for fill systems in permeable soils with shallow creviced or porous bedrock. Small Scale Waste Management Project, Univ. of Wisconsin, Madison, WI, 24 pp.

Converse, J. C., R. J. Otis and J. Bouma. 1976c. Design and construction procedures for fill systems in permeable soils with high water tables. Small Scale Waste Management Project, Univ. of Wisconsin, Madison, WI, 24 pp.

Cotlove, E., V. Trautham and R. L. Bowman. 1958. An instrument for a method for automatic, rapid, accurate and sensitive titration of chlorides in biological samples. J. Lab. Clin. Med. 50:358-371.

Delfino, J. J. 1977. Contamination of potable ground water supplies in rural areas. *In* Pojasek, R. B. (ed.), Drinking Water Quality Enhancement through Source Protection. Ann Arbor Science Pub. Ind., Ann Arbor, MI, pp. 275-295.

Deutsch, H. F. 1979. Pesonal communication. Professor of Physiological Chemistry, Univ. of Wisconsin, Madison.

Dudley, J. G. and D. A. Stephenson. 1973. Nutrient enrichment of ground water from septic tank disposal systems. Upper Great Lakes Regional Commission, University of Wisconsin-Extension, Madison, WI., 131 pp.

Ellis, B. and K. E. Childs. 1973. Nutrient movement from septic tank and lawn fertilization. Tech. Bull. No. 73-5. Dept. of Natural Resources, Lansing, MI. 83 pp.

Foth, H. D. and L. M. Turk. 1972. Fundamentals of Soil Science. 5th Ed. John Wiley and Sons, New York, 453 pp.

Gerba, C. P., E. M Smith and J. L. Melnick. 1977. Development of a quantitative method for detecting enteroviruses in estuarine sediments. Appl. Environ. Microbiol. 34:158-163.

Gerba, C. P., C. Wallis and J. L. Melnick. 1975. Fate of wastewater bacteria and viruses in soil. J. Irrig. Drain Div., Amer. Soc. Agr. Engs. 101:157-174.

Gibbs, M. M. 1977. Soil renovation of effluent from a septic tank on a lakeshore. N.Z. J. Sci. 20:255-263.

Gruener, N. and N. I. Shuval. 1970. Health aspects of nitrates in drinking water. *In* N. I. Shuval (ed.), Developments of Water Quality Research. Ann Arbor Science, Ann Arbor, MI, pp. 127-141.

Harkin, J. M. 1977. Method for treating septic tank effluent seepage beds and the like. U.S. Pat. 4,021,338. Assigned to the Wisconsin Alumni Research Foundation, Madison, Wis. POROX is a registered trademark of WARF.

Harkin, J. M. 1979. Presentation at 4th Annual Program on On-site Treatment and Disposal of Small Wastewater Flows. University of Wisconsin-Extension, March, 1979.

Harkin, J. M., M. D. Jawson and F. G. Baker. 1976. Causes and remedy of failure of septic tank seepage systems. Proc. Second National Conference on Individual Onsite Wastewater Systems, National Sanitation Foundation, Ann Arbor, MI, pp. 119-124.

Hetling, L. J. and I. G. Carcich. 1973. Phosphorus in wastewater. Water Sew. Works 120:59-62.

Hickey, J. L. S. and D. L. Damien. 1966. Performance of single family septic tank systems in Alaska. J. Water Pollut. Control Fed. 38:1298-1309.

Hill, D. E. 1972. Wastewater renovation in Connecticut soils. J. Environ. Qual. 1:163-169.

Hoadley, D. 1976. Potential health hazards associated with *Pseudomonas aeruginosa* in water. *In* D. Hoadley and W. Dutka (eds.), Bacterial Indicators/Health Hazards Associated with Water. Amer. Soc. Testing Mats. Philadelphia, PA., Pub. No. STP 635. pp. 8-114.

IES. 1979. Water resources workshop--Water management plan for Fourteen Mile Creek watershed, Adams Co., Wisconsin. Report #105, Institute for Environmental Studies, Univ. of Wisconsin, Madison, WI. 388 pp.

Iskandar, S. 1979. Personal communication. Research Associate, Dept. Soil Science, Univ. of Wisconsin-Madison.

Jawson, M. 1976. Soil clogging by septic tank effluent: Causes and methods of correction. M.S. Thesis, Univ. of Wisconsin, Madison, WI. 138 pp.

Jones, R. A. and G. F. Lee. 1977. Septic tank disposal systems as phosphorus sources for surface waters. EPA Report 600/2-77/129, Washington, D.C. 74 pp.

Kao, C. V. and R. W. Blanchar. 1973. Distribution and chemistry of phosphorus in an albaqualf soil after 82 years of phosphate fertilization. J. Environ. Qual. 2:237-240.

Keeney, D. R. and W. R. Gardner. 1975. The dynamics of nitrogen transformations in the soil. *In* F. Singer (ed.), The Changing Global Environment. D. Reidel Pub., Dordrecht, Holland. pp. 367-375.

Keller, W. D. and G. E. Smith. 1967. Ground-water contamination by dissolved nitrate. Geol. Soc. Amer. Spec. Pap. 90, pp. 47-59. Kriese, M. 1974. Methemoglobinemia: A Comprehensive Treatise. Chemical Rubber Co. Press, Cleveland, OH.

Laak, R. 1970. Influence of domestic wastewater pretreatment on soil clogging. J. Water Pollut. Control Fed. 42:1495-1500.

Laak, R. 1976. Pollutant loads from plumbing fixtures and pretreatment to control soil clogging. J. Environ. Health 39(1):48-51.

Magdoff, F. R., J. Bouma and D. R. Keeney. 1974a. Columns, representing mound type disposal systems for septic tank effluent. I. Soil-water and gas relationships. J. Environ. Qual. 3:223-228.

Magdoff, F. R., D. R. Keeney, J. Bouma and W. A. Ziebell. 1974b. Columns representing mound-type disposal for septic tank effluent. II. Nutrient transformations and bacterial populations. J. Environ. Qual. 3:228-234.

Magdoff, F. R. and D. R. Keeney. 1976. Nutrient mass balance in columns representing fill systems for disposal of septic tank effluents. Environ. Letts. 10:285-294.

Maynard, D. N., A. V. Barker, P. L. Minotti and N. H. Peck. 1976. Nitrate accumulation in vegetables. Advan. Agron. 28:71-118.

McCoy, E. and W. A. Ziebell. 1975. The effects of effluents on groundwater: bacteriological aspects. *In* Proc. Second National Conference on Individual On-site Wastewater Systems, National Sanitation Foundation, Ann Arbor, MI, pp. 67-76.

McKinney, N. and A. Krause. 1979. Seven lakes examined in EIS: "Septic snooper" plan will save billions of dollars. Environment Midwest, June 1979, pp. 12-15.

Miller, R. 1978. Preliminary environmental impact statement for three alternate systems (mounds) for on-site individual wastewater disposal in Wisconsin. Dept. Health Social Serv., State of Wisconsin, Madison, WI, 220 pp.

Mitchell, R. 1972. Water Pollution Microbiology. John Wiley and Sons, Inc., New York. 416 pp.

Otis, R. J., J. Bouma and W. G. Walker. 1974. Uniform distribution in soil absorption fields. Ground Water 12:409-417.

Otis, R. J., W. C. Boyle and D. K. Sauer. 1975. Performance of household wastewater treatment units under field conditions. *In* Proc. Natl. Home Sewage Disposal Symp., Amer. Soc. Agr. Engrs., St. Joseph, MI, pp. 191-201.

Patrick, W. H. and K. R. Reddy. 1974. Effect of alternate aerobic and anaerobic conditions on redox potential, organic matter decomposition and nitrogen loss in a flooded soil. J. Soil Biol. Biochem. 7:87-94.

Patrick, W. H. and K. R. Reddy. 1976. Effect of frequent changes in aerobic and anaerobic conditions on redox potential and nitrogen loss in flooded soil. J. Soil Biol. Biochem. 8:491-495.

Peavy, H. S. and K. S. Groves. 1977. The influence of septic tank drain fields on groundwater quality in areas of high groundwater. *In* Proc. Sec. Natl. Home Sewage Treatment Symp., Amer. Soc. Agr. Engs., St. Joseph, MI, pp. 218-225.

Polkowski, L. B. and W. C. Boyle. 1970. Ground water quality adjacent to (a) septic tank-soil absorption system. State of Wisconsin Dept. of Natural Resources, Madison, WI. 75 pp.

Pruel, H. C. 1967. Underground movement of nitrogen. Adv. Water Poll. Res. Vol. 1, pp. 309-329. Water Poll. Contr. Fed., Washington, D.C.

Quan, E. L., H. R. Sweet and J. R. Illian. 1974. Subsurface sewage disposal and contamination of ground water in East Portland, Oregon. Ground Water 12(6):356-367.

Riddel, W. E., F. W. Oehme and D. C. Kelley. 1974. Nitrates in Kansas groundwater as related to animal and human health. Ecology 2:397-405.

Sauer, D. K., W. C. Boyle and R. J. Otis. 1976. Intermittent sand filtration of household wastewater under field conditions. J. Environ. Eng. Div., Amer. Soc. Civ. Engs. 108:789-803.

Sawhney, B. L. 1977. Predicting phosphate movement through soil columns. J. Environ. Qual. 6:86-89.

Sawyer, C. N. 1965. Problems of phosphorus in water supplies. J. Amer. Water Works Assoc. 57:1431-1442.

Schindler, D. W. Evolution of phosphorus limitation in lakes. Science 195:260-262.

Schuknecht, R., G. W. Lawton, P. Steinke and J. J. Delfino. 1975. Nitrates in Wisconsin groundwater. Environ. Letts. 9(1):91-98.

Schutt, K., J. C. Converse, D. Valla and R. J. Otis. 1979. Locating and landscaping the mound system of on-site waste-water disposal. Small Scale Waste Management Project, Univ. of Wisconsin, Madison, WI, 12 pp.

Shannon, E. E. 1975. Effects of detergent formulation on wastewater characteristics and treatment. J. Water Pollut. Control Fed. 47:2371-2382

Shannon, E. E. and P. L. Brezonik. 1972. Relationship between lake trophic state and nitrogen and phosphorus loading rates. Environ. Sci. Technol. 6(8):719-724.

Sherman, H. C. 1952. Chemistry of Food and Nutrition. 8th Ed. MacMillan Co., New York. 479 pp.

Shuval, H. I. and N. Gruener. 1972. Epidemiological and toxicological aspects of nitrates and nitrites in the environment. Amer. J. Publ. Health 62(8):1045-1052.

Siegrist, R., M. W. Witt and W. C. Boyle. 1976. The characteristics of rural household wastewater. J. Environ. Engr. Div., Amer. Soc. Civ. Engs. 102:533-548.

Sikora, L. J. and D. R. Keeney. 1975. Laboratory studies on simulation of biological denitrification. *In* Proc. Natl. Home Sewage Disposal Symp., Amer. Soc. Agr. Engs., St. Joseph, MI, pp. 38-41.

Sikora, L. J. and R. B. Corey. 1976. Fate of nitrogen and phosphorus in soils under septic tank waste disposal systems. Amer. Soc. Agr. Engr. 19:866-875.

Sikora, L. J., M. G. Bent, R. B. Corey and D. R. Keeney. 1976. Septic nitrogen and phosphorus removal test system. Ground Water 14:309-314.

Simons, A. P. and F. R. Magdoff. 1979a. Disposal of septic tank efluent in mound and sand filter-trench systems on a clay soil. J. Environ. Qual. 8(4):469-473.

Simons, A. P. and F. R. Magdoff. 1979b. Laboratory evaluation of design parameters for mound system disposal of septic tank effluent. J. Environ. Qual. 8(4):486-492.

Soil Science Society of America. 1975. Glossary of soil science terms. Soil Sci. Soc. Amer., Madison, WI. 34 pp.

Standard Methods for the Examination of Water and Wastewater. 1975. 14th Ed., American Public Health Association, Washington, D.C., 968 pp.

Standard Specifications for Highway Material and Methods of Sampling and Testing. 1955. Standard methods of mechanical analysis of soils. AASHO Desig. T88-54:257-266.

Tabatabai, M. A. and J. M. Bremner. 1970. Use of the Leco automatic 70 second carbon analyzer for total carbon analysis of soils. Soil Sci. Soc. Amer., Proc. 34:608-610.

Tofflemire, T. J. and M. Chen. 1977. Phosphate removal by sands and soil. Ground Water 15:377-387.

Tyler, E. J., R. Laak, E. McCoy and S. S. Sandhu. 1977. The soil as a treatment system. *In* Proc. Second Natl. Home Sewage Treatment Symp., Amer. Soc. Agr. Engs., St. Joseph, MI, pp. 22-37.

U.S. Dept. of Agriculture. 1975. Soil Taxonomy. Soil Conserv. Serv., U.S.D.A. Handbook No. 436, Washington, D.C.

U.S. Geological Survey. 1976. Mineral and water resources of Wisconsin. U.S. Govt. Printing Office, Washington, D.C.

Viraraghavan, T. and R. G. Warnock. 1976a. Efficiency of a septic tile system. J. Water Poll. Contr. Fed. 48:934-944.

Viraraghavan, T. and R. G. Warnock. 1976b. Water quality adjacent to a septic tank system. J. Amer. Water Works Assoc. 68(11):611-614.

Walker, W. G., Jr. 1973. Nitrogen transformation during subsurface disposal of septic tank effluent in sands. M.S. Thesis, Univ. of Wisconsin-Madison.

Walker, W. G., J. Bouma, D. R. Keeney and F. R. Magdoff. 1973a. Nitrogen transformations during subsurface disposal of septic tank effluent in sands. I. Soil transformations. J. Environ. Qual. 2(4):475-480.

Walker, W. G., J. Bouma, D. R. Keeney and P. G. Olcott. 1973b. Nitrogen transformations during subsurface disposal of septic tank effluent in sands. II. Ground water quality. J. Environ. Qual. 2(4):521-525.

Walker, W. H. 1973. Ground-water nitrate pollution in rural areas. Ground Water 11(5):19-22.

Wendt, R. 1974. Phosphorus determinations in soils using $SrCl_2$ extraction. M.S. Thesis, Univ. of Wisconsin-Madison.

West, E. S., W. R. Todd, H. S. Mason and J. T. Van Burgen. 1966. Textbook of Biochemistry. 4th Ed. McMillan, New York.

Winton, E. F., R. G. Tardiff and L. J. McCabe. 1971. Nitrate in drinking water. J. Amer. Water Works Assoc. 63:95-98.

Wisconsin State Board of Health. 1969. Administrative Code H62.20. State Bd. Health, State of Wisconsin. Revised Aug. 1976.

Witt, M. W., R. Siegrist and W. C. Boyle. 1975. Rural household characteristics. *In* Proc. Home Sewage Disposal Symp. Amer. Soc. Agr. Engs., St. Joseph, MI, pp. 79-88.

Woodward, F. L., F. J. Kilpatrick and P. B. Johnson. 1961. Experiences with ground water contamination in unsewered areas in Minnesota. Amer. J. Publ. Health 51(8):1130-1136.

Witz, R. L. 1975. Twenty-five years with the Nodak waste disposal system. *In* Proc. Natl. Home Sewage Disposal Symp., Amer. Soc. Agr. Engr. St. Joseph, MI., pp.168-174.

Young, J. L. and M. R. Lindbeck. 1964. Carbon determination in soils and organic materials with a high-frequency induction furnace. Soil Sci. Soc. Am. Proc. 22:377-380.

Ziebell, W. A., D. H. Nero, J. T. Deininger and E. McCoy. 1975. Fecal bacteria: Removal from sewage by soils. *In* Proc. Natl. Home Sewage Disposal Symp., Amer. Soc. Agr. Engs., St. Joseph, MI, pp. 58-63.